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Proceedings of the 7th Annual Research Symposium Held at Philadelphia, Pennsylvania on March 16-18, 1981

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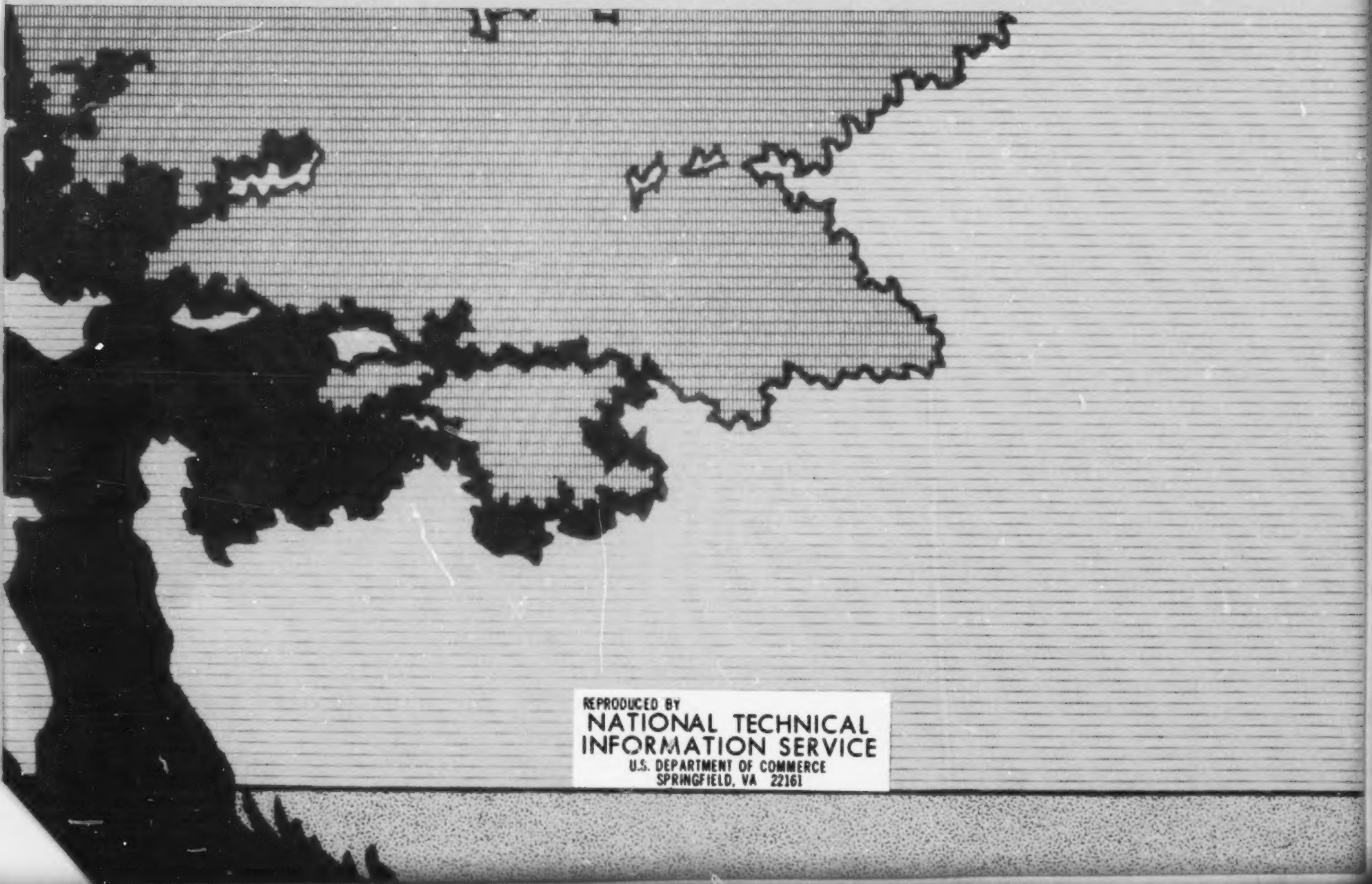

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16. ABSTRACT

The Seventh Annual SHWRD Research Symposium on land disposal of municipal solid waste and industrial solid waste and resource recovery of municipal solid waste was held in Philadelphia, Pennsylvania, on March 16, 17, and 18, 1981. The purposs of the symposium were (1) to provide a forum for a state-of-the-art review and discussion of on-going and recently completed research projects dealing with the management of solid and industrial wastes; (2) to bring together people concerned with municipal solid waste management who can benefit from an exchange of ideas and information; and (3) to provide an arena for the peer review of SHWRD's overall research program. These proceedings are a compilation of papers presented by the symposium speakers. The technical areas covered in the Land Disposal: Municipal Solid Waste are gas and leachate production, treatment and control technologies and economics. The areas covered in Land Disposal: Hazardous Wastes are hazardous waste characterization, transport and fate of pollutants, hazardous waste containment, land treatment of hazardous wastes, hazardous waste treatment, uncontrolled sites/remedial action, and economics. Municipal Solid Waste: Resource Recovery include the areas of equipment and processing, recovery and use of materials, environmental aspects and economics/impediments and special studies.

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FOREWORD

The Environmental Protection Agency was created because of increasing public and governmental concern about the dangers of pollution to the health and welfare of the American people. Noxious air, foul water, and spoiled land are tragic testimony to the deterioration of our natural environment. The complexity of the environment and the interplay between its components require a concentrated and integrated attack on the problem.

Research and development is the first necessary step in problem solution; it involves defining the problem, measuring its impact, and searching for solutions. The Municipal Environmental Research Laboratory develops new and improved technology and systems to prevent, treat, and manage wastewater and the solid and hazardous waste pollutant discharges from municipal and community sources; to preserve and treat public drinking water supplies; and to minimize the adverse economic, social, health and aesthetic effects of pollution. This publication is one of the products of that research--a vital communications link between the researcher and the user community.

These Proceedings present the results of completed and ongoing research projects concerning the land disposal of municipal solid waste.

Francis T. Mayo
Director
Municipal Environmental
Research Laboratory

PREFACE

These Proceedings are intended to disseminate up-to-date information on extramural research projects concerning the land disposal of municipal solid wastes. These projects are funded by the Solid and Hazardous Waste Research Division (SHWRD) of the U.S. Environmental Protection Agency, Municipal Environmental Research Laboratory in Cincinnati, Ohio.

The papers in these Proceedings are arranged as they were presented at the symposium and have been printed basically as received from the authors. They do not necessarily reflect the policies and opinions of the U.S. Environmental Protection Agency. Hopefully, these Proceedings will prove useful and beneficial to the scientific community as a current reference on the land disposal of municipal solid wastes.

ABSTRACT

The Seventh Annual SHWRD Research Symposium on land disposal of municipal solid waste, hazardous waste, and resource recovery of municipal solid waste was held in Philadelphia, Pennsylvania, on March 16, 17, and 18, 1981. The objectives of the symposium were (1) to provide a forum for a state-of-the-art review and discussion of ongoing and recently completed research projects dealing with the management of solid and hazardous wastes; (2) to bring together people concerned with municipal solid waste management who can benefit from an exchange of ideas and information; and (3) to provide an arena for the peer review of SHWRD's overall research program. These proceedings are a compilation of papers presented by the symposium speakers.

The symposium proceedings are being published as three separate documents. In this document, Land Disposal: Municipal Solid Waste, three technical areas are covered. They are as follows:

(1) Gas and leachate production
(2) Treatment and control technologies
(3) Economics

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CURRENT RESEARCH ON LAND DISPOSAL OF MUNICIPAL SOLID WASTES

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ABSTRACT

The Solid and Hazardous Waste Research Division (SHWRD), Municipal Environmental Research Laboratory, U.S. Environmental Protection Agency, in Cincinnati, Ohio, has responsibility for research in the areas of municipal solid and hazardous waste management, including both disposal and processing. This research is being directed towards new and improved systems of municipal solid and hazardous waste management, development of technology, determination of environmental effects, and collection of data necessary for the establishment of disposal and processing guidelines.

Division activities in the area of municipal solid waste research have in the past related to storage, collection, transport, processing, resource recovery, disposal, and waste-as-fuels programs. However, over the years the municipal solid waste research has been de-emphasized. As a result storage, collection and transport research activities have been non-existent for several years. Recently, resource recovery, waste-as-fuels and disposal have been de-emphasized so that in FY 1981 no new funds will be available for these research areas.

The current municipal solid waste disposal research program has been divided into three general areas: (1) Pollutant Predictions for Current Landfill Techniques, (2) Alternatives to Current Landfill Disposal Techniques, and (3) Remedial Action for Minimizing Pollutants from Unacceptable Sites.

The research activities currently funded under these three general areas have been classified into five categories shown below:

1. Leachate Forecasting
2. Controlled Decomposition
3. Co-Disposal
4. Manuals of Practice
5. Economic Assessment

INTRODUCTION

Increasing amounts of waste residuals are being directed to the land of disposal in landfills. At the same time, there is increasing evidence of environmental damage resulting from improper operation. The burden of operating landfills and coping with any resulting damages falls most heavily on municipalities and other local government agencies. Their problems are complex, involving legislation, economics, and public attitudes as well as technology; additionally, comprehensive information on landfilling techniques and protection of the local environment is not readily available.

The municipal waste disposal program initiated by SHWRD was designed to document and evaluate the potential adverse environmental and public health effects that could result from application of waste disposal methods without proper precautions for leachate and gas management. This research strategy, encompassing state-of-the-art documents, laboratory analysis, bench and pilot studies, and full-scale field verification studies, is at various stages of implementation. The waste disposal research program was intended to develop and compile a research criteria data base for use in the development of guidelines and standards for waste residual disposal to the land as mandated by the recently enacted legislation entitled "Resource Conservation and Recovery Act of 1976" (RCRA). This intention was actively pursued through FY 1980. At this time, it was decided that other research priorities (i.e., hazardous wastes) required additional financial resources and this resulted in a de-emphasis of the municipal solid waste (MSW) land disposal program. This de-emphasis reduced the funding for MSW land disposal to zero in FY 1981 and FY 1982. As a result of this action, no new research starts are planned and only no cost continuation research activities are being pursued. The current research program in this area will terminate in FY 1982. As a consequence, the intention of developing guidance documents for MSW residuals disposal has not been totally achieved. Some best engineering judgement documents have been developed as relates to pollutant generation and control, but a total wrap-up and summation of all aspects of MSW land disposal technology will not be achieved. Certain areas as co-disposal, leachate treatment, gas production, migration and control are still largely unexplored.

LEACHATE FORECASTING

The objective of this program is to develop a technique to predict the volumetric production of leachate and to predict the quality or composition of leachate from selected wastes. Achievement of this objective will allow the necessary input data to assessment of natural soil/leachate interactions, an assessment of potential environmental and public health adverse effects, and scheduling and selection of control processes to avoid unreasonable adverse effects.

One recently completed effort (1) is the monitoring of simulated landfill test cells constructed at a research field installation of the US EPA at Walton, Kentucky. This field site known as the "Boone County Field Site" (BCFS) was constructed to evaluate the pollution potential associated with the sanitary landfill method of municipal solid waste dipsosal. Leachate and gas sampling and analysis, temperature and data has been collected and published in an interim report entitled "Boone County Field Site Interim Report, Tests Cells 2A, 2B, 2C and 2D", EPA-600/2-79-058. A final report will be published in 1981 and will include results from all the simulated test cells.

A second completed effort (1) compiled a description and critical analysis of landfill leachate information to advance the state-of-the-art for management of landfill leachates from municipal solid wastes. The report will provide practical information on the generation, significance, and costs for controlling of landfill leachates for use by design engineers and landfill operators. This report will be published in the near future after technical reviews are completed.

Several other on-going and completed efforts (1) relating to leachate and gas migration and movement are discussed in another paper being given at this symposium entitled "Current Research on Land Disposal of Hazardous Wastes".

CONTROLLED DECOMPOSITION

The objective of this program is to develop techniques which may be used to control the quality and volume of leachate production. Quality control is achieved by slowing or accelerating the decomposition processes. Volumetric control of leachate production is achieved by improving infiltration control, and development of liner and leachate treatment technology.

Waste Leachability

The objective of the initial effort (1) which is completed, was to confirm laboratory studies of the leachate recycle concept with larger prototype test cells and to elucidate information on mass flux of gas and leachate components with particular attention to the effect of evapotranspiration on the rates and quantities of leachate.

Two doubly-lined test cells received identical volumes of water and weight of shredded solid waste. Gas was monitored for CO_2, CH_4, other constituents and total volume. Leachate was monitored for BOD, COD, TOC, total and individual volatile acids, alkalinity, acidity, pH, nitrogen and phosphorus, chlorides, sulfides or sulfates, and pertinent heavy and alkaline earth metals.

Mass flux was determined for the contaminant monitored and analyses made to interpret and control the stabilization process. Recommended design and control procedures for leachate containment and recycle was then developed. An economic assessment was also performed. The final report will be published in 1981.

A second completed effort (1) investigated the effects of preprocessing municipal refuse prior to landfilling. The preprocessing procedures considered were shredding, baling, and a combination of both techniques. Leachate volume and composition of the gases produced were assessed. The study was accomplished through the use of landfill simulators buried in the ground. The final report will be published in 1981.

A third completed effort (1) investigated the health and environmental significance of the persistence of fecal strepococci found in leachate from landfilled municipal refuse. This study was carried out in two phases.

The initial phase verified microbial analytical methods and determined the presence of study organisms in a variety of leachates. Samples of leachate from different sources of landfilled waste representing different stages of waste decomposition and different operational conditions were assayed for microbial and chemical content. Microbial assays included total aerobic and anaerobic plate counts, indicators of fecal pollution, selected bacterial pathogens and anaerobes and the major fungi of pathogenic significance. Chemical analyses were used to describe the leachate environment.

The second phase investigated the relationship between the extent of waste decomposition and the microbial population dynamics. Three experimental lysimeters containing municipal refuse, municipal refuse and sewage sludge, and hospital waste were constructed at 16" of net infiltration per year. This final report will be published in 1981.

The objective of the fourth effort (1) which is currently on-going, is to assess the impact of leachate from a sanitary landfill on a conventional activated sludge process for treatment of municipal wastewater. The evaluation will emphasize the effect of leachate on process performance and effluent quality and its effect on the economics of process operation.

Two parallel conventional activated sludge processes are being operated at a nominal flow of 20 gpm and the influent is effluent from the primary settling basins at the Mill Creek Sewage Treatment Plant. Leachate is added to the influent flow of one activated sludge process, the other process functions as the experimental control.

During the first phase of the project the biological processes were operated at low sludge ages such that nitrification is not established. The second phase of the project will be similar to the procedure followed during the first phase except sludge ages in both systems will be increased in order to maintain complete nitrification.

The fifth effort (2) which is completed, evaluated the aesthetic factors associated with land disposal of milled solid waste in a variety of particle sizes without daily cover.

Four experimental test plots were constructed containing four different particle size distributions. The variables evaluated were:

- effect of wind velocities on refuse displacement
- differential settlement
- initial density and effects of time
- vegetative growth
- vectors and wildlife.

The results of this study will be published in a final report entitled

"Effects of Particle Size on Landfilled Solid Waste: Cold Climate Studies" in 1981.

Liners/Membranes/Admixtures

The liner/membrane/admixture technology is being studied to evaluate suitability for eliminating or reducing leachate from landfill sites containing municipal wastes.

The initial effort (3) for liner materials being investigated under the municipal solid waste program included six admixed materials and six flexible membranes. The admixed materials were:

- o 2 asphalt concretes, varying in permeability
- o 1 soil asphalt
- o 2 asphalt membranes, one based on an emulsified asphalt and the other on catalytically-blown asphalt
- o 1 soil cement

The six flexible membranes are:

- o Butyl rubber
- o Ethylene propylene rubber (EPDM)
- o Chlorinated polyethylene (CPE)
- o Chlorosulfonated polyethylene (HYPALON)
- o Polyethylene (PE)
- o Polyvinyl chloride (PVC)

The results of the 1-year exposure have been discussed in a report entitled "Evaluation of Liner Materials Exposed to Leachate - Second Interim Report", EPA-600/2-76-255, September 1976. The exposure of these liner materials has been extended to a 5 month period. Data on gas migration, water permeability (as in soils), osmotic, and finger print analysis was collected for individual polymeric materials. Interim results were published in a report entitled "Liner Materials Exposed to Leachate - Third Interim Report", EPA-600/2-79-038. The final results of this completed effort will be published in 1981 in a report entitled "Evaluation of Liner Materials Exposed to Leachate - Final Report".

In the second completed effort (3), field verification of liners was undertaken. The objective of this task was to obtain samples of lining materials from existing sanitary landfills and to evaluate and compare the physical properties of the sample specimens to original specifications so that length of service may be estimated.

Permission to utilize four landfills was obtained. Information compiled on these landfills includes: length of time since closure of each portion, type of lining material, relative amounts of municipal and non-municipal solid waste placed in the landfill, and methods of operation.

In the testing and evaluation portion of this effort, changes in the physical properties of the lining materials were determined and in the cases of selected soil and admixed materials, chemical changes with depth were determined. The results of this effort are being prepared into a report for publication in 1981.

A third effort (3), currently underway, will assess preplacement procedures used in constructing lined facilities. Actual field procedures utilized in a) preparing the supporting structure and b) placing the various liner materials common to projects requiring positive control of fluid loss will be assessed.

Subgrade preparation and liner placement activities have been observed at fifteen sites to date. These sites include landfills, wastewater impoundments, and potable water reservoirs.

CO-DISPOSAL

The overall objective of the co-disposal activity is to assess the impact of the disposal of industrial waste materials with municipal solid waste. Concern has been voiced that the addition of industrial waste may result in the occurrence of various toxic elements in leachate thereby posing a threat to potable groundwater supplies. Because the environmental effects from landfilling result from not only the soluble and slowly soluble materials placed in the landfill but also the products of chemical and microbiological transformation, these transformations should be a consideration in management of a landfill to the extent that they can be predicted or influenced by disposal operations.

One completed effort (1) reviewed methods for disposal of Wastewater Treatment Plant (WTP) residual by landfilling. Nitrate nitrogen, total dissolved solids, and trace metals appear to represent the greatest pollution hazard to water resources for nearly all types of residuals. Co-disposal (WTP sludge/refuse) techniques were found to be the most cost effective. The WTP sludge-only trench disposal and co-disposal methods appear to be the most environmentally acceptable. This report is being finalized for publication in 1981.

A second effort (1) is currently underway to determine the fate of heavy metals released from industrial sludges in municipal refuse during landfill decomposition stabilization. Leachate recirculation is being used to accelerate the degradation processes. An evaluation of this leachate recirculation will be made.

A third on-going effort (1) is a study of factors influencing the effect of admixing industrial sludges and sewage sludge with municipal solid wastes. A combination of municipal solid waste and various solid and semi-solid industrial wastes along with municipal digested primary sewage sludge were added to several field lysimeters. All material flows were measured and characterized for the continuing study and related to leachate quality and quantity, gas production and microbial activity. A report describing the results from the activity will be published in 1981.

A fourth effort (4) examined potential effects of co-disposal using municipal solid waste landfill leachate to extract leachate pollutants from industrial wastes and study the leachate movement in soil columns. A sequential batch leaching and soil adsorption procedure was developed to evaluate soil type reactivity to attenuate leached contaminants. These results will be published in a report in 1981.

MANUALS OF PRACTICE

The initial effort (5), which is completed, investigated the environmental impact of special types of landfills. Volume reduction (millfill and balefill) and alternate methods of waste disposal (hillfill and strip mine landfill) were evaluated and compared with standard sanitary landfill methods. The relative environmental impacts of these methods on water resources were determined along with site characteristics that contribute to or prevent pollution. Projections were also made for the probable usefulness of each method in geographical areas other than those studied. The results of this effort will be reported in 1981 in a final report publication entitled "Environmental Impacts of Special Types of Landfills".

A second effort (5) recently completed, was undertaken to determine: a) if current sampling methods produce samples that are representative of water contained in the aquifer being monitored; b) if groundwater samples collected in the field must be treated on location or if they can be brought back to the laboratory for treatment without altering their chemical nature; and c) which sampling and preservation techniques should be accepted as standards for groundwater sampling.

Six landfill monitoring wells were studied using four different pumping techniques and thirteen different sample preservation procedures. Chemical analyses of samples from each pumping method and preservation procedure were conducted for fifteen constituents.

The results of this completed project will be published in 1981 in a final report entitled "Groundwater Sampling and Sample Preservation Techniques".

The third effort (3) recently completed was undertaken to determine which tree species can best maintain themselves in a landfill environment; to investigate the feasibility of preventing landfill gas from penetrating the root zone of selected species by using gas barrier techniques; and to identify those factors which are most important in maintaining adequate plant growth on completed sanitary landfills. Ten replicates of nineteen woody species were planted on a ten year old completed sanitary landfill and five gas barrier systems were constructed. Of the nineteen species planted, black gum proved most tolerant and honey locust least tolerant to anaerobic landfill conditions. Of the five gas barrier systems tested, three proved effective in preventing penetration of gas into the root systems of the tested species.

The results of this study are currently being prepared for presentation into a report for publication in 1981.

ECONOMIC ASSESSMENT

The overall objective of this research activity is to assess the considerations involved in solid waste disposal and the various alternatives available within an economic framework.

One on-going effort (6) is evaluating the relative importance of the various factors affecting the cost of sanitary landfilling of solid waste and providing methods for evaluating the cost of combinations of waste processing and sanitary landfilling. This will identify the least cost disposal alternative under local conditions.

CONCLUSIONS

The laboratory and field research project efforts discussed here reflect the overall SHWRD effort in municipal solid waste disposal research. The projects will be discussed in detail in the following papers. More information about a specific project or study can be obtained by contacting the Project Officer referenced in the text. Inquiries can also be directed to the Director, Solid and Hazardous Waste Research Division, Municipal Environmental Research Laboratory, U.S. Environmental Protection Agency, 26 West St. Clair Street, Cincinnati, Ohio 45268. Information will be provided with the understanding that it is from research in progress and that conclusions may change as techniques are improved and more complete data become available.

PROJECT OFFICERS

All the Project Officers can be contacted through the Solid and Hazardous Waste Research Division (SHWRD), whose address is shown above. The telephone number for all Project Officers is: 513/684-7871.

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LEACHATE PRODUCTION AND MANAGEMENT FROM MUNICIPAL LANDFILLS: SUMMARY AND ASSESSMENT

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ABSTRACT

This report is a review, analysis, and evaluation of current literature and information on municipal landfill leachates. The overall objective is to provide documentation of practical information, current techniques, comparative costs, and additional research needs on the generation, composition, migration, control, and monitoring of leachates from municipal landfills. Due to the sizeable amount of information available on the subject matter only the conclusions are presented in this report.

INTRODUCTION

A major environmental impact resulting from the disposal of municipal solid waste into a sanitary landfill occurs when water passing through the waste accumulates various contaminants. This percolate or leachate may enter surrounding surface waters or underlying groundwaters and seriously degrade the water quality. An extensive body of literature is available which describes the phenomenon responsible for municipal landfill leachate generation, composition, migration, control, and monitoring. The objectives of this study are to (1) clarify the understanding of leachate production and management through critical analysis of existing information, (2) develop practical information, techniques, and comparative costs which can be used by design engineers and site operators, and (3) identify areas where additional research may be expected to yield practical information. This information is presented in terms of five subject areas:

- Leachate generation;
- Leachate composition;
- Leachate migration;
- Leachate control; and
- Environmental monitoring.

LEACHATE GENERATION

Factors Affecting Leachate Generation

In general, leachate volume generation is determined by the following four conditions:

- Availability of water;
- Landfill surface conditions;
- Refuse conditions; and
- Characteristics of the underlying soil.

The major factors which affect the above conditions and subsequently affect the leachate generation are summarized in Figure 1.

Factors affecting the availability of water include direct precipitation, surface runoff, groundwater intrusion, irrigation, water from refuse decomposition, and compaction of liquid waste or sludge co-disposal refuses. Of these water sources, primary contributor is direct precipitation. Precipitation in adjacent areas can be channeled onto the landfill site. If refuse is placed in or just above the groundwater table, a groundwater mound may saturate the refuse. Considerable quantities of leachate will result. Irrigation applied

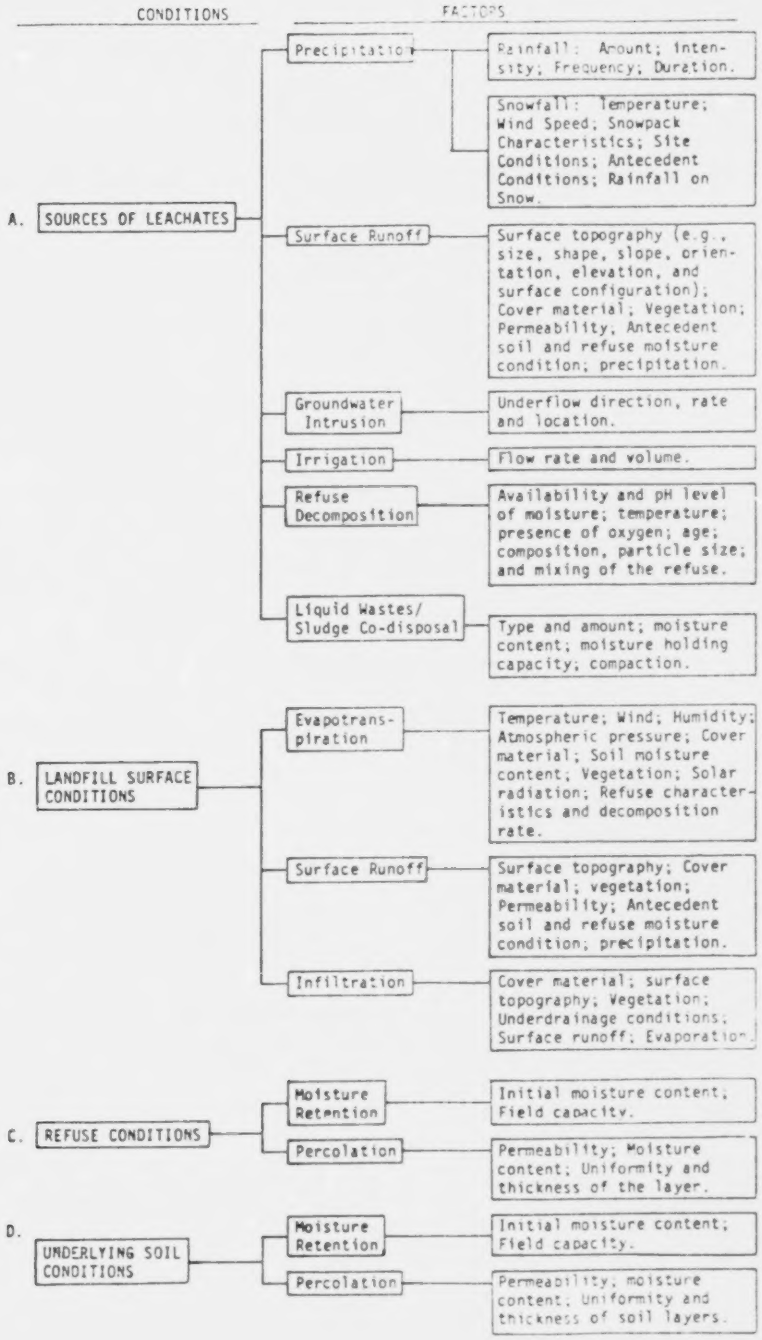
CONDITIONS
FACTORS
A. SOURCES OF LEACHATES
Precipitation
Rainfall: Amount; Intensity; Frequency; Duration.
Snowfall: Temperature; Wind Speed; Snowpack Characteristics; Site Conditions; Antecedent Conditions; Rainfall on Snow.
Surface Runoff
Surface topography (e.g., size, shape, slope, orientation, elevation, and surface configuration); Cover material; Vegetation; Permeability; Antecedent soil and refuse moisture condition; precipitation.
Groundwater Intrusion
Underflow direction, rate and location.
Irrigation
Flow rate and volume.
Refuse Decomposition
Availability and pH level of moisture; temperature; presence of oxygen; age; composition, particle size; and mixing of the refuse.
Liquid Wastes/ Sludge Co-disposal
Type and amount; moisture content; moisture holding capacity; compaction.
B. LANDFILL SURFACE CONDITIONS
Evapotranspiration
Temperature; Wind; Humidity; Atmospheric pressure; Cover material; Soil moisture content; Vegetation; Solar radiation; Refuse characteristics and decomposition rate.
Surface Runoff
Surface topography; Cover material; vegetation; Permeability; Antecedent soil and refuse moisture condition; precipitation.
Infiltration
Cover material; surface topography; Vegetation; Underdrainage conditions; Surface runoff; Evaporation.
C. REFUSE CONDITIONS
Moisture Retention
Initial moisture content; Field capacity.
Percolation
Permeability; Moisture content; Uniformity and thickness of the layer.
D. UNDERLYING SOIL CONDITIONS
Moisture Retention
Initial moisture content; Field capacity.
Percolation
Permeability; moisture content; Uniformity and thickness of soil layers.

Figure 1. Factors affecting leachate generation

to the landfill surface may also contribute to leachate generation. This is especially true if the completed landfill is used for recreational purposes. Water from refuse decomposition is another potential contributor. Co-disposal of liquid wastes or sludge to municipal landfills may also contribute to leachate generation.

Water reaching the landfill surface may either evaporate or transpire, infiltrate through the landfill surface, or leave the site as surface runoff. Water entering the cover soil may only change the soil to field capacity (field capacity is defined as the maximum moisture content which a soil or a solid material can retain in a gravitational field without producing continuous downward percolation) during transmittal. Pathways available to the water and the distribution of water among them will depend principally upon the landfill surface conditions. Surface conditions which may affect leachate generation include vegetation, soil permeability, cover material (type, dimension, compaction, etc.), surface topography, slope, temperature, humidity, and wind speed above the landfill.

Once field capacity of the surface cover material is attained, leachate will percolate through the refuse. Retention and transmission characteristics of the refuse control the percolation rate. Leachate may be channelled through the refuse and dispersed by refuse intermediate cover layers, or may seep through the pores of the refuse. If channelling does not occur leachate will not be produced in a landfill until at least a portion of the refuse reaches field capacity. Any additional moisture will then cause leachate movement. Underlying soil conditions can modify both the rate and amount of leachate generation when soils underlying and surrounding the site have lower permeabilities than the cover soils and refuse.

The quantity of leachate produced may also differ considerably with management or operating practices, depending on whether the leachate is viewed as a long- or short-term problem (1). Operational factors may include cover material handling, watering prior to compaction, daily variation in compaction and cell construction, and variation in waste composition (e.g. municipal refuse plus sludge or industrial wastes; milled or unmilled refuse).

Concepts and Techniques Describing Leachate Generation

Numerous mathematical methods have been used for quantitative estimation of the volume of leachate generated from landfills (7, 9,12,13,16,21,22). Although these approaches vary, they rely upon water balance or water budget principles. The water balance method is based on: (1) a one-dimensional flow model and conservation of mass relationships among various components of the leachate sources, and (2) the retention and transmission characteristics of the refuse and cover soil. The generalized water balance relationships are shown in Table 1 and illustrated in Figure 2.

Equation (1) in Table 1 illustrates the water balance relationship of the landfill surface. Water contributed by precipitation (W_p), surface runoff (W_{SR}), or irrigation (W_{IR}) will either become surface runoff (R), or infiltrate into the cover soil (I). A portion of the infiltrated water leaves by evapotranspiration

TABLE 1.

GENERALIZED WATER BALANCE EQUATION AT A

MUNICIPAL LANDFILL SITE

$$W_P + W_{SR} + W_{IR} = I + R \quad \ldots\ldots (1)$$

where: W_P = Input water from precipitation
W_{SR} = Input water from surrounding surface runoff
W_{IR} = Input water from irrigation
I = Infiltration
R = Surface runoff

$$PER_S = I - E - \Delta S_S \quad \ldots\ldots (2)$$

$$PER_R = I - E - \Delta S_S + W_D - \Delta S_R \quad \ldots\ldots (3)$$

$$= PER_S + W_D - \Delta S_R$$

where: PER_S and PER_R = Percolation in soil and refuse respectively
W_D = Water contributed by solid waste decomposition
ΔS_S = Change in moisture storage in soil
ΔS_R = Change in moisture storage in refuse
E = Evapotranspiration

and $$L = PER_R + W_{GW} \quad \ldots\ldots (4)$$

where: L = Leachate generation
W_{GW} = Input water from underflow

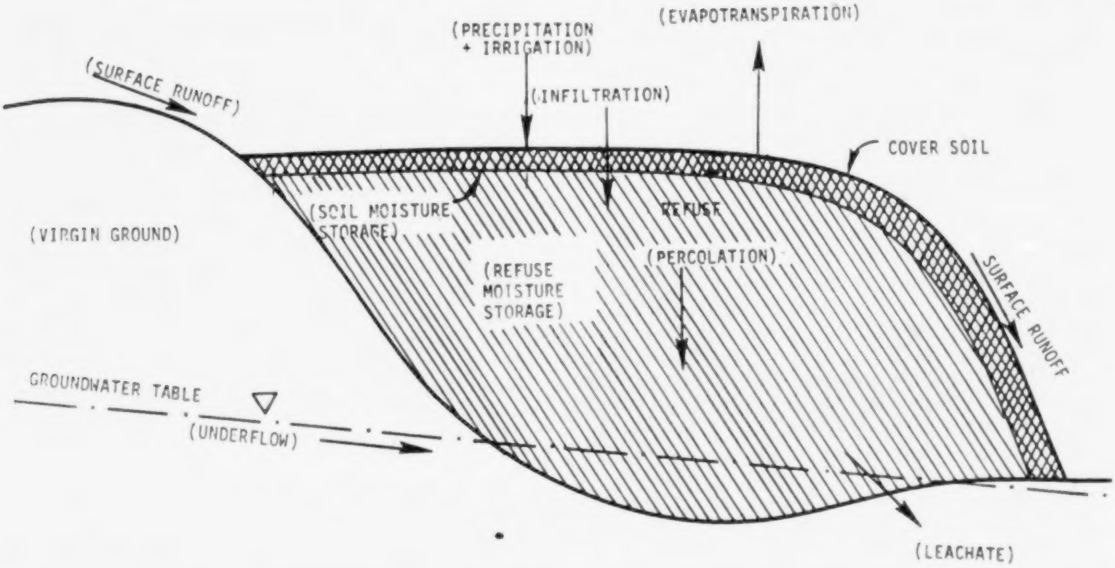

Figure 2. Municipal landfill water balance

while a portion will recharge the cover soil. Once the field capacity of the cover soil is attained, vertical percolation (PER_S) will occur. The quantity of percolation through the cover material can be calculated by using equation (2). At first the percolate will be absorbed by the refuse. As the landfill refuse reaches field capacity, refuse percolate (PER_R) is generated in quantities described by equation (3). The percolate will eventually evolve as refuse leachate (L). If groundwater intrusion (W_{GW}) occurs, leachate generation estimates can be modified using equation (4).

Because leachate generation mechanisms are site specific, any estimation method requires knowledge of site topography, geology, hydrology, climatology, and meteorology. The many poorly known factors in the water balance equations often require empirical, rational, experimental methods. These different approaches reflect the wide variety of concepts describing leachate generation. In this report, only those techniques which are appropriate for municipal landfills are discussed.

Figure 3 shows the techniques applicable for landfill leachate volume estimation. As shown by this figure, a combination of two hundred and forty (240) different methods are available for leachate generation calculation. Data needed for each method is listed in the first column of Figure 3.

The calculation (Figure 3) involves two major steps: (1) quantification of leachate generation factors (precipitation, surface runoff, infiltration, etc.) and (2) water balance calculations. Details of the calculation procedures are presented in the original report (11).

Discussion

The practicality and accuracy of the leachate generation techniques were evaluated by comparing calculated results with field leachate measurements. Five municipal landfill sites representing different geographical areas and site conditions were selected. Because of the incomplete data available (e.g. some evaluations require lysimeter or field testing data), this study evaluated only twenty-five (25) methods. They consist of combinations of the following techniques:

- Precipitation: field measurements;
- Surface runoff: rational method, and curve number method;

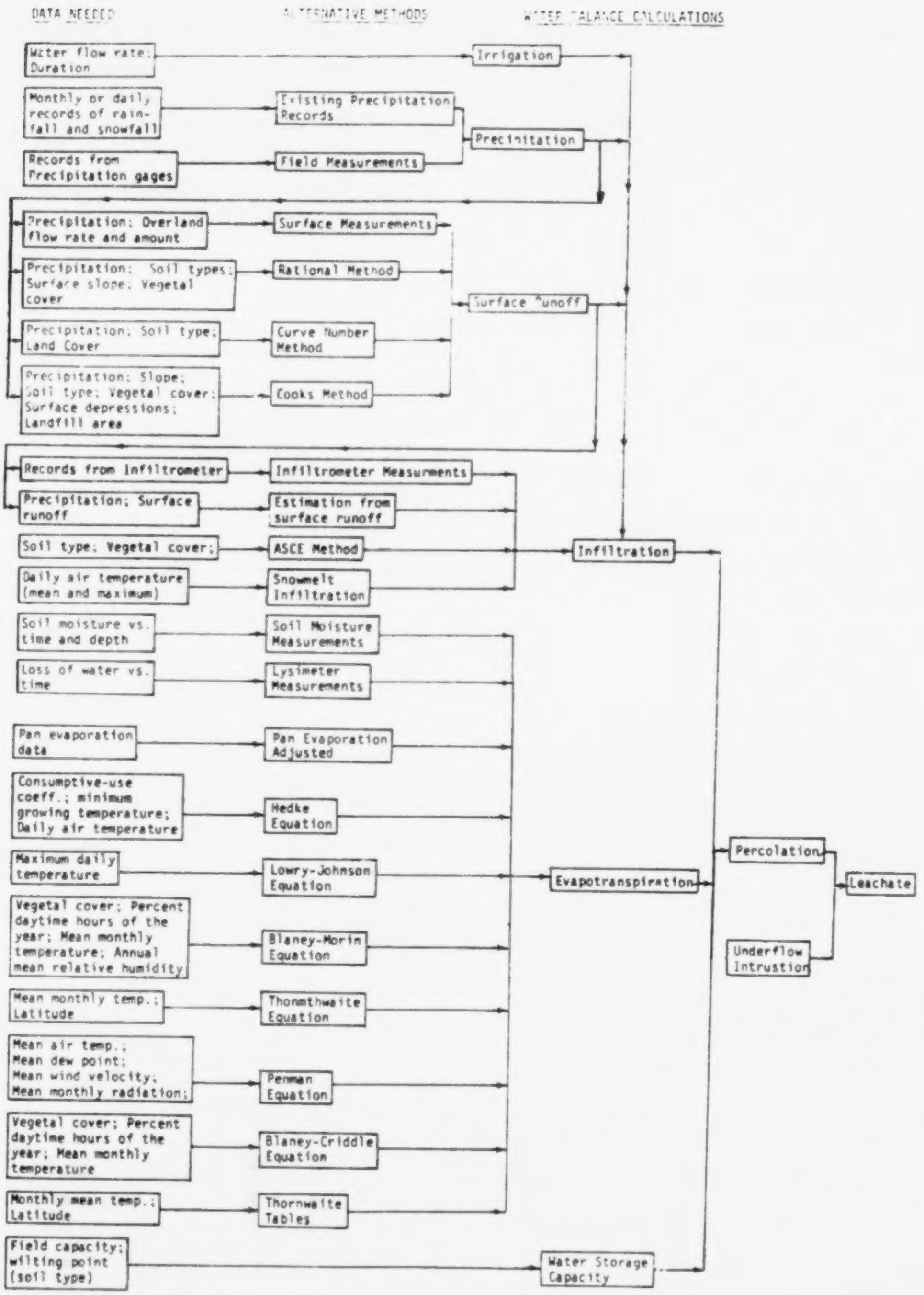
DATA NEEDED
ALTERNATIVE METHODS
WATER BALANCE CALCULATIONS
Water flow rate; Duration
Irrigation
Monthly or daily records of rainfall and snowfall
Existing Precipitation Records
Records from Precipitation gages
Field Measurements
Precipitation
Precipitation; Overland flow rate and amount
Surface Measurements
Precipitation: Soil types; Surface slope; Vegetal cover
Rational Method
Surface Runoff
Precipitation; Soil type; Land Cover
Curve Number Method
Precipitation; Slope; Soil type; Vegetal cover; Surface depressions; Landfill area
Cooks Method
Records from Infiltrometer
Infiltrometer Measurments
Precipitation; Surface runoff
Estimation from surface runoff
Soil type; Vegetal cover;
ASCE Method
Infiltration
Daily air temperature (mean and maximum)
Snowmelt Infiltration
Soil moisture vs. time and depth
Soil Moisture Measurements
Loss of water vs. time
Lysimeter Measurements
Pan evaporation data
Pan Evaporation Adjusted
Consumptive-use coeff.; minimum growing temperature; Daily air temperature
Hedke Equation
Maximum daily temperature
Lowry-Johnson Equation
Evapotranspiration
Percolation
Leachate
Vegetal cover; Percent daytime hours of the year; Mean monthly temperature; Annual mean relative humidity
Blaney-Morin Equation
Underflow Intrustion
Mean monthly temp.; Latitude
Thornthwaite Equation
Mean air temp.; Mean dew point; Mean wind velocity; Mean monthly radiation;
Penman Equation
Vegetal cover; Percent daytime hours of the year; Mean monthly temperature
Blaney-Criddle Equation
Monthly mean temp.; Latitude
Thornwaite Tables
Field capacity; wilting point (soil type)
Water Storage Capacity

Figure 3. Flow chart of waste balance calculation

- Infiltration: estimation from surface runoff, and ASCE method; and
- Evapotranspiration: pan evaporation adjusted, Blaney-Morin equation, Blaney-Criddle equation, Thorthwaite equation, and Thornthwaite table.

Evaluation results show that accuracies of the methods range from 1.32 percent to 5389 percent, depending on the models used and the site conditions selected. Among the 125 individual cases (25 methods x 5 sites) studied, 54 cases (about 43 percent) are underestimated for leachate generation, and 71 cases are overestimated.

No one method estimated leachate generation with reasonable accuracy for various site conditions. The following methods are found to achieve better leachate estimation (less than 250% of average accuracy):

- Use Perry's table for runoff estimation, calculate infiltration from surface runoff, and use the Blaney-Morin equation for evapotranspiration estimation;
- Use the Salvato table for runoff estimation, calculate infiltration from surface runoff, and use the Blaney-Criddle equation for evapotranspiration estimation;
- Use the ASCE rational method for runoff estimation, calculate infiltration from surface runoff, and use the Thornthwaite equation for evapotranspiration estimation;
- Use Perry's table for runoff estimation, calculate infiltration from runoff, and use the Thornthwaite Tables for evapotranspiration estimation;
- Use Perry's table for runoff estimation, calculate infiltration from runoff, and use the Blaney-Morin equation for evapotranspiration estimation;

The results suggest the following methods may achieve less than 100 percent error for the listed areas:

- Perry's rational method: northwest, midwest, and south;
- Salvato rational method: northwest, midwest, and south;
- Thornthwaite method (equation or table): east and midwest;
- ASCE rational method: east;
- ASCE infiltraton method: northwest and west;
- Pan evaporation method: midwest and south;
- Blaney-Morin method: northwest and midwest; and
- Blaney-Criddle method: northwest and west.

However, the described comparisons are based on limited field data. Part of the error may actually result from field monitoring: gathering representative leachate from a landfill is difficult. No landfills were designed specifically for the purpose of evaluating leachate generation models. Complete information for the purpose of leachate generation estimation is not available. For example, the runoff coefficients used in these example calculations were selected from tables used for other engineering or agricultural purposes and may not be suitable for landfill surface conditions. In order to accurately assess leachate production models,additional research designed for landfills is required.

LEACHATE COMPOSITION

Chemical and Microbiological Composition

Figure 4 illustrates concentration ranges of leachate chemical composition results obtained through a comprehensive literature survey. As shown by Figure 4, concentration levels of leachate constituents vary considerably. They may range from trace levels for certain parameters to 90,000 mg/l for others. Identical constituents in different tests also demonstrated variations in the concentration levels of up to four orders of magnitude difference.

Several classes of organic compounds have been identified in landfill leachates (2, 3, 4, 8,10,17,18). In general, they can be classified into three groups: (1) fatty acids of low molecular weight, (2) humic, carbohydrate-like substances of high molecular weight, and (3) fulvic-like substances of intermediate molecular weight Researches have revealed that organic constituents in landfill leachates are largely a factor of landfill age. For relatively unstabilized landfills, results

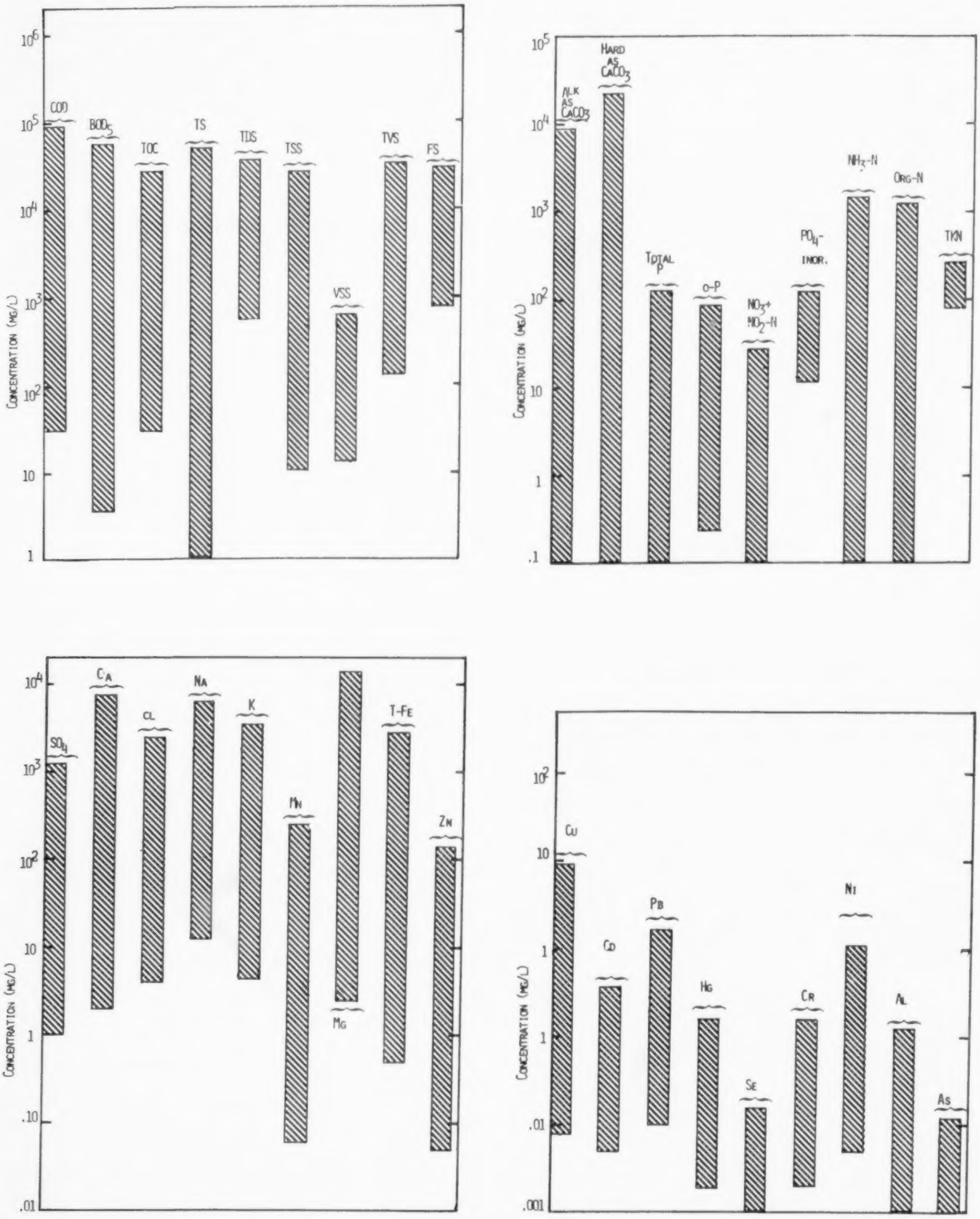

FIGURE 4 . RANGE OF MUNICIPAL LANDFILL LEACHATE CONSTITUENT CONCENTRATIONS

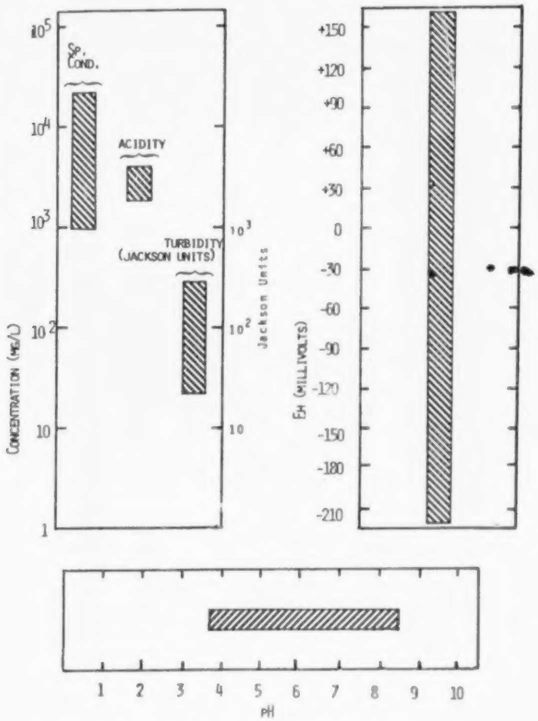

FIGURE 4 (CONCLUDED)

show that up to ninety (90) percent of the soluble organic carbon can be accounted for as short-chain volatile fatty acids; acetic, propionic, and butyric acids being present in greatest concentrations. The next largest fraction was usually fulvic acids with a relatively high density of carboxyl and aromatic hydroxyl groups. For relatively stabilized landfills, volatile fatty acids were decreased over time and fulvic-like fractions were increased.

Little information exists concerning the microbiological composition and the survival of pathogenic microorganisms contained in landfill leachates. Bacteria most frequently isolated from the landfills are species of the genera Bacillus, Corynebacterium, and Streptococcus. Other common bacteria are species of Achromobacter, Acinetobacter, Aeronomonas, Clostridium, Enterobacter, Listeria, Micrococcus, Moraxella, Neisseria, Pseudomonas, Serratia, and Staphylococcus. Fungi isolated from the experimental landfill leachates include Allescheria boydii, Cephalosporium, sp., Fusarium sp., Penicillium sp., Sepedonium sp., and yeast. Very little information is available in the literature on viruses present in landfill leachates. Enteric viruses have been identified in leachates from several landfills. Reports of other viruses found in landfill leachates are not conclusive.

Factors Affecting Leachate Composition

Leachate composition is a function of numerous factors including those inherent in the refuse mass and landfill location, and those created by engineers and site operators. The physical, chemical, and biological properties of refuse are largely uncontrollable, although shredding of refuse changes leachate composition to some degree. Similarily ambient air temperatures and rainfall are unalterable characteristics of a landfill site. Conversely, the rate of water application, refuse permeability, refuse depth, and temperature within the refuse bed can be regulated.

Among the factors evaluated, landfill age is found to be the most related factor affecting leachate composition in the field condition. Existing data shows that other factors may have relationships with leachate composition only under the controlled conditions (e.g. when evaluating one factor, other factors are constant). A first-order rate equation finds application for expression of the relationships between landfill age and upper or lower concentration boundaries of leachate composition. Figures 5 to 8 are examples of such relationships.

Leachate Composition Models

The complex nature of landfill stabilization and the interaction of landfill design and operational variables create a highly unpredictable leachate character. The quantitative prediction of leachate composition and contaminant flux in a municipal landfill has received little attention. Wigh (23) suggested an empirical equation to express the leachate concentration based on cumulative leachate volume. The equation is applicable only to test cells in which cumulative leachate volume can be easily measured. Qasim and Burchinal (14) developed a concept which

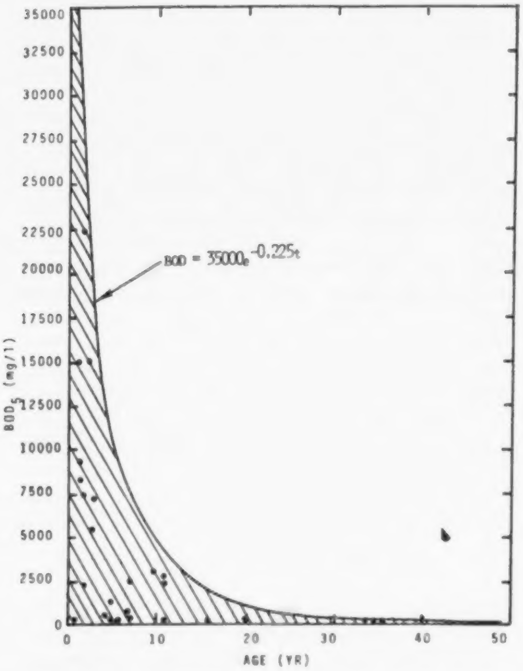

Figure 5. Landfill Age vs. Leachate BOD_5

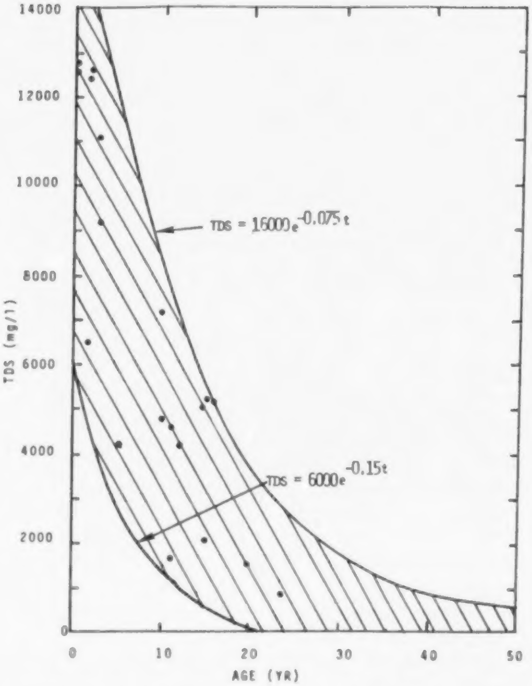

FIGURE 6. Landfill Age vs. Leachate TDS

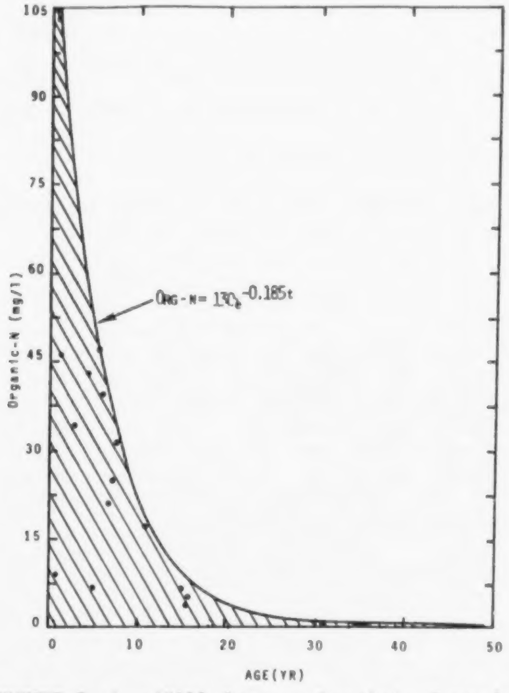

FIGURE 7. Landfill Age vs. Leachate Organic Nitrogen Concentration

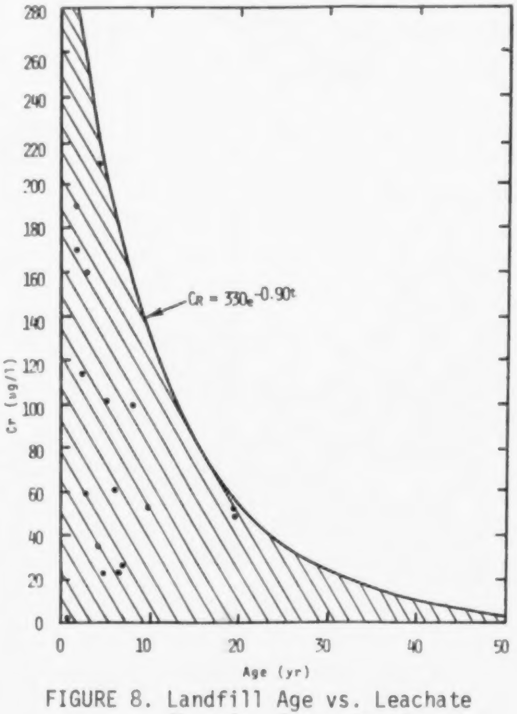

FIGURE 8. Landfill Age vs. Leachate Chromium Concentration

assumed chloride to be a surrogate parameter of landfill leachate composition. Comparison of the empirical data with theoretical estimates were accurate during the initial study phase, but underestimated chloride concentrations as leaching continued.

In this study, empirical equations based on landfill age were developed to estimate leachate composition. The equations differ from previous modeling efforts in that leachate concentration histories from actual field experiments are described rather than batch type laboratory and pilot scale experiments. Due to the complexity of the leachate composition and incompleteness of the data, only concentration maxima or minima (e.g. boundaries) could be solved. Results of equations and rate constants are shown in Table 2. However, this approach failed to indicate trends for concentration boundaries for nitrite, nitrate, phosphate, and some trace metals. Additional concentration history data may reveal empirical relationships and appropriate factors representing leachate composition.

TABLE 2.
RATE EQUATIONS AND RATE CONSTANTS FOR LEACHATE CONSTITUENT CONCENTRATION HISTORIES

Rate Equation	Units	Rate Constant, k	Reaction Order
$BOD_5 = 35000e^{-0.225t}$	mg/l	0.225	1
$COD = 89000e^{-0.192t}$	mg/l	0.192	1
$TOC = 14000e^{-0.26t}$	mg/l	0.26	1
$TVS = 24000e^{-0.185t}$	mg/l	0.185	1
$SP\text{-}COND = 20850e^{-0.10t}$	mg/l	0.10	1
$TDS = 16000e^{-0.075t}$	mg/l	0.075	1
$Org\text{-}N = 130e^{-0.185t}$	mg/l	0.185	1
$NH_4\text{-}N = 12000e^{-0.10t}$	mg/l	0.10	1
NO_3, NO_2 -N = n.a.$^{+}$	n.a.	n.a.	n.a.
Total P = n.a.	n.a.	n.a	n.a.
$SO_4 = 15000e^{-0.079t}$	mg/l	0.079	1
$AlK.^{*} = 12000e^{-0.072t}$	mg/l	0.072	1
$Cl = 2470e^{-0.065t}$	mg/l	0.065	1
$Ca = 24000e^{-0.074t}$	mg/l	0.074	1
Na = n.a.	n.a.	n.a	
$Cr = 330e^{-0.90t}$	μg/l	0.90	1
$Cu = 10e^{-0.20t}$	mg/l	0.20	1
$Cd = 160e^{-0.125t}$	μg/l	0.125	1
Fe = n.a.	n.a	n.a	n.a.
Mn = n.a.	n.a.	n.a.	n.a.
Pb = n.a.	n.a	n.a	n.a.
Zn = n.a.	n.a.	n.a.	n.a.
pH = n.a.	n.a.	n.a	n.a

* as $CaCO_3$

+ not available

LEACHATE MIGRATION

The multitude of reactions that are operative when contaminants enter a soil can be qualitatively identified with reasonable assurance, but quantitative data relating to specific mechanisms are not available. Considerable effort has been placed on the development of sophisticated leachate migration models (19). However, due to the highly complex leachate/soil environments, no leachate migration model exists that can simulate all of the physical, chemical, and biological processes occurring in a typical landfill system. In the following pages, conclusions of the leachate migration studies are presented. The original contract report provides detailed discussion of these studies.

Leachate/Environment Interactions

Soil Properties--

The soil properties most useful in predicting mobility of leachate contaminants are: (1) texture (clay content); (2) content of hydrous oxides (Fe, Mn, and Al); and (3) type and content of organic matter. Relative mobilities of leachate contaminants through soils are quite variable. However, a qualitative prediction is possible if the important soil physical and chemical properties are known.

Migration Mechanisms--

The most important physical properties of a soil system relative to leachate migration are diffusion and dispersion, dilution, physical sorption, and straining. Chemical mechanisms which can regulate the leachate/soil interactions include precipitation/dissolution, sorption, complexation, ion exchange, and redox reactions. Biological reactions affecting contaminant migration are numerous. The following major effects can be attributed to microbial activities: redox effects, mineralization, immobilization, precipitation/dissolution, and complexation.

Migration Trends of Contaminants--

General parameters--General parameters studied include pH, redox potential, and alkalinity. pH levels in leachates are usually acidic because of biodegradation reactions. Low leachate pH levels can usually be increased when leachate percolates through the underlying soils, mainly by the dissolution of carbonate, oxide, or sulfide solids. The redox condition of leachates from active biodegradation landfills is usually reducing. Soluble sulfides produced in the landfill is the main species which controls the redox level. In the leachate/soil environment the fate of alkalinity is mainly controlled by the precipitation/dissolution of calcite in the soil.

Organics--Bioconversion, surface sorption, and metal-organic complexation were found to be the major mechanisms for the migration of organics in the leachate/soil system. These factors may either increase or decrease the organics in leachates. Extensive data on the fate of organic matter in leachate/soil systems indicate that the migration of organic matter is largely site specific.

Major ions--Possible mechanisms for regulating the levels of major ions are solubilization and ion-exchange effects. In leachate/soil systems, chloride, and sodium were found to be relatively unattenuated, while calcium and potassium were moderately attenuated.

Nutrients--In the leachate/soil system, adsorption is considered one of the most important mechanisms for controlling movement of ammonia-nitrogen. The nitrate-nitrogen is relatively mobile and not retained by ion-exchange processes. The most important mechanisms governing the phosphorus migration are solubilization, sorption, and biological effects. The soluble phosphate levels in the neutral leachates are most likely controlled by $Ca_5OH\,(PO_4)_3(s)$ and regulated by complexation ions. Because of this phenomenon, the calcium ion concentration or alkalinity level in leachates will be the dominant factor to affect the soluble phosphate levels.

Trace Metals--The movement of trace metals in the leachate/soil system is extremely complex. Not only are there numerous controlling factors, but many unknowns beyond present state-of-the-art knowledge are involved. The most important mechanisms that influence mobility of trace metals are solubilization, sorption, complexation, and dilution. Because of these reactions, metal concentrations in leachates may either be increased or decreased through the underlying soils.

Chlorinated hydrocarbons--Sorption is believed to be the most significant factor to regulate the levels of chlorinated hydrocarbons in the leachate/soil system. Clay minerals, iron and manganese hydrated oxides, and organics are responsible for the decrease of chlorinated hydrocarbons in leachates.

Migration Models

Models available for predicting leachate migration can be classified as descriptive, physical, analog, and mathematical (19). For each of these four types of models, the following distinctions are found: empirical or conceptual, stochastic or deterministic, static or dynamic, and spacial dimensionality of the model. Among the above varieties, the conceptual-mathematical models appear to be the most effective for leachate migration prediction. Such models are currently available with differences between the models arising from the number of simplifications made during the derivation of the basic equations (mass transport and flow equations) method of solving the equations, and stipulated boundary conditions. Over a hundred of such mathematical models are available from the literature. However, as yet, no model exists that can simulate all of the physical, chemical, and biological processes that are operative in a typical waste disposal system.

LEACHATE CONTROL

Technologies available for municipal leachate control can be classified into the following four types:

- Leachate volume control;
- Leachate composition control;
- Leachate collecton; and
- Leachate treatment and ultimate disposal.

Leachate Volume Control

Leachate volume control refers to the control of leaching fluids into or out of the fill. Two approaches are generally practiced: groundwater control and surface water control. Available control techniques include the following:

- Groundwater control measures
  - (1) Groundwater/leachate isolation
    - (i) Artificial liners
    - (ii) Bottom sealers
    - (iii) Slurry trench
    - (iv) Grout curtain
    - (v) Sheet piling cutoff wall
  - (2) Leachate plume control
    - (i) Drains
    - (ii) Well point system
    - (iii) Deep well system
- Surface water control
  - (1) Contour grading and surface water diversion
  - (2) Surface sealing
  - (3) Revegetation (Land Stabilization)

Tables 3 to 5 summarize the suitable site conditions, limitations, costs, and perceived effectiveness for the use of the above control methods.

Leachate Composition Control

The major objective of leachate composition control is to reduce the strength

TABLE 3. GROUNDWATER CONTROL MEASURES

Control Technique	Optimum Site Conditions	Limitations	Costs
Artificial Liner	Groundwater near to the refuse/soil interface. Underlying soils possess a moderately rapid to very rapid permeability 1.4×10^{-3} cm/sec to $>1.4 \times 10^{-2}$ cm/sec.	Requires installation prior to landfilling. Expected life not established. Rising groundwater may rupture the liner. May be damaged during landfilling activities Once installed, extremely difficult to repair	\$1.00 to \$5.00 installed/yd^2
Bottom Sealer	Same as for artificial liner and when a detailed understanding of the soil is known.	Detailed understanding of soil properties are required. Only practical prior to landfilling.	\$1.20 to \$2.00 per installed yd^2
Slurry Trench	Near to surface groundwater and bedrock condition. Other conditions similar to those for an artificial liner.	Costs of shipping bentonite. Excavation difficult due to soil conditions. Deterioration of bentonite when exposed to high strength leachate may occur.	\$294 to \$495 per installed linear foot.
Grout Curtain	Same as for an artificial liner as well as a near to surface bedrock condition	Considered effective in soils with permeabilities $>10^{-5}$ cm/sec. Costs may be prohibitive for large scale use. Difficult to assess the integrity of the seal.	\$142 to \$357 per installed [illegible]^3
Sheet Piling Cutoff Wall	Same as for grout curtain.	Corrosion potential and problems with driving the sheet through rocky soils, ability to maintain integrity of the piling.	\$382 to \$562 per installed linear ft.

TABLE 4. LEACHATE PLUME CONTROL

Control Technique	Optimum Site Conditions	Limitations	Economics	Perceived Effectiveness
Drains	Near-to-surface ground water; soils which are easily excavated, consistent ground water level with little fluctuation.	Groundwater may be too low for effective use of drains. Soils may not be amenable to evcavation. Area required for drainage may be too large an area for practical drainage purposes.	\$300-\$800/acre	Effective when used in conjunction with other control approaches such as liners, well points, etc.
Well point system	Same as for drains	Usually effective for shallow dewatering purposes ($\leq$10 m)	Capital cost, \$1.33 per lineal ft. Maintenance and operational costs; \$1.67 per lineal ft.	Effective when used to compliment another control system.
Deep well system	A greater quanity of water exists than a well point system may withdraw.	Capable of withdrawing approximately 40 vertical ft. of water in uniform sands.	Capital cost \$18,000 to dewater 900 linear ft. Maintenance and operational costs \$10,500/year.	Effective in radically the hydraulic gradient of a large body of water in a small area.

TABLE 5. SURFACE WATER CONTROL

Control Techniques	Optimum Site Conditions	Limitations	Economics	Perceived Effectiveness
Contour Grading and Surface Water Diversion	Landfill stabilized; sufficient barrow area for soil excavation available nearby.	Available soil cover should be present. Differential settlement is extensive requiring large volumes of soil.	$126,000 to $242,000 for a 10 acre landfill.	An effective control measure used both during landfilling and once the fill has been stabilized.
Surface Sealing	Borrow material available; landfill has been stabilized and/or graded.	Suitable cover material may not be readily available. Gas venting may also be required.	$140,000 to $256,200 for a 10 acre site*	Effective if settlement is minimal and when used in conjunction with other control measures.
Revegetation	Stabilized fill and available cover soil (> 2 ft.). Minimum of settlement and gas control equipment available at the site.	Decomposition gases may be toxic to vegetation if not vented or controlled.	.12¢ to .18¢ per acre.	Provides a buffer to moisture entering the fill, and reduces erosion. Is most effective when employed in conjunction with several control measures.

* Based for a clay cap 6" thick; assumes earthmoving costs per acre at $1,000.

and contaminant flux of leachate. This can be done through:

- Control of refuse stabilization rates by construction and operation features;
- Leachate recirculation; or
- Addition of municipal sludge, industrial wastes, or selected sorbents.

Landfill construction and operation features which are significant in influencing leachate composition are the physical characteristics of refuse, including particle size (shredding) and density (compaction and baling); rate of water application; landfill depth or lift height; and landfill temperature (which can be regulated to some extent through cover material, refuse density, and lift height). In general, higher water application rate, smaller particle size, and higher temperature increase the refuse stabilization rates, and therefore, favor the reduction of leachate composition. However, the effectiveness, application range, costs, and practicality of employing this method is less studied.

Leachate recirculation offers several advantages including the natural acceleration of refuse stabilization and the absence of leachate treatment systems. Cited disadvantages are the high capital and maintenance costs, and the need to carefully monitor the leachate application. Three types of leachate recirculation processes are recognized: spray irrigation, overland flow, and subgrade irrigation. The suitable site conditions, limitations, economics, and perceived effectiveness of these techniques are illustrated in Table 6.

Co-disposal of dewatered sewage sludge with municipal refuse can increase the refuse stabilization because of pH buffer, increased microbial population, and moisture content. Disadvantages include the potential for pathogens, organics, and metals in leachate, and odor problems.

Receipt of alkaline and nutrient bearing industurial wastes may accelerate stabilization and provide control over leachate composition. A distinct lack of data is evident for this co-disposal technique.

Laboratory studies have demonstrated the effectiveness of various natural and synthetic materials in removing contaminants from leachates. Selected materials included bottom ash, fly ash, vermiculite, illite, ottowa sand, activated carbon, Kaolinite, natural zeolites, activated alumina, and cullite. The results have

TABLE 6. LEACHATE RECIRCULATION AND COLLECTION TECHNOLOGIES

Control Techniques	Optimum Site Conditions	Limitations	Economics	Perceived Effectiveness
Leachate Recirculation				
Spray Irrigation	High evaporation rates for the site; site vegetation well established. Site contouring minimizes ponding runoff, etc.	Practical only in dry months; wet weather conditions minimizes effectiveness. Pumping equipment, collection wells and associated piping is required.	Capital $30,900 (solid set). Operational and maintenance costs $3,500/year.	Most effective in arid regions or during the summer months when treatment efficiency is the highest.
Overland Flow	Land availability with existing leachate collection, spray equipment, and ditches for recycle: existence of rooted grasses and slope conducive to natural gravity flow. Favorable soil properties and high evaporation/evapotranspiration rates desired.	Large area requirements; leachate collection and distribution system required conducive weather conditions required for optimum use.		Provides an effective treatment scheme when land is available and climatic conditions are favorable.
Subgrade Irrigation	Collection and leachate distribution system exists for the fill.	Elaborate piping and distribution system required	$500 to $3,000 per acre Operational and maintenance costs ≈ $1,500/year	Provides an effective landfill stabilizatin mechanism ; largely unaffected by climatic conditions. Treatment efficiency is less than for spray irrigation or overland flow.
Leachate Collection Systems				
Shallow Drains	Shallow bedrock condition; homogenous off-site soils	For most effective use, require installation prior to landfilling; difficult to repair once damaged.	$300 to $800 per acre	Represents an effective collection method when properly designed.
Shallow Well Point System	Near-to-surface groundwater conditions	Electrical costs will increase; continuous use of equipment necessary for optimum use	See Leachate Plume Control	See Leachate Plume Control

indicated that no single sorbent of those examined can significantly reduce the concentration of all constituents in leachate to acceptable levels. The results, however, indicate that sorbent combinations can be used to reduce all the contaminants to acceptable levels. More pilot or field studies are necessary to identify the practicality, limitations, costs, and effectiveness of this technique.

Leachate Collection

Shallow interceptor drains and well systems are most commonly used for leachate collection. The equipment and techniques are identical to that described for leachate volume control. Table 6 gives the suitable site conditions, limitations, economics and perceived effectiveness of these techniques.

Leachate Treatment and Ultimate Disposal

Some researches (1, 5) have shown that an extended aeration activated sludge plant can accept up to five (5) percent by volume of leachate without seriously impairing effluent quality. Others (1,15) have questioned the ability of treatment plants to remove the variety of chemicals found in leachate. A number of studies have been conducted which examine the treatment efficiency of unit processes designated specifically for landfill leachate. Chain and DeWalle (6) performed an extensive literature survey dealing with treatment methods for landfill leachates. They concluded that newly formed landfill leachate is best treated by aerobic or anaerobic biological treatment processes. Leachate from stabilized landfills is best treated with physical-chemical treatment processes.

Figure 9 summarizes the COD removal efficiencies for various demonstrated unit treatment systems. Among the methods studied, the trickling filter, the aerated lagoon, and anaerobic digestion processes appear to be most effective. Substantial reduction in inorganic contaminants generally cannot be achieved when utilizing those processes as described previously (except anaerobic filter). Figure 10 indicates those treatment alternatives that have been tried for ammonia and heavy metals. Air stripping and activated

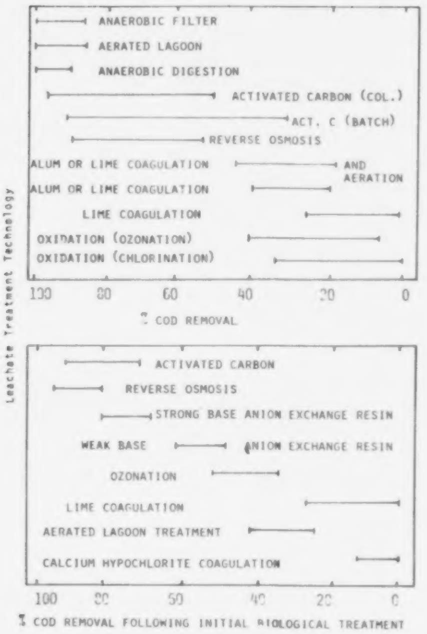

Figure 9. COD removal efficiencies for the treatment of raw leachates

sludge show high reduction in ammonia concentrations, while lime coagulation appears to be the optimal alternative for heavy metal removal. Combinations of chemical/physical, and biological processes were found most effective in treating a variety of landfill leachates. Field scale treatment facilities are seldom employed and studied. Additional research is therefore suggested.

LEACHATE MONITORING

The Environmental Protection Agency has published a manual titled "Procedures Manual for Groundwater Monitoring at Solid Waste Disposal Facilities" (EPA/530/SW-611) which provides a detailed guidance for leachate monitoring. This study complements the material presented in the manual with further information which is useful to landfill designers and site operators. Three topic areas are included:

- Monitoring in the zone of aeration;
- Monitoring in the zone of saturation; and
- Approaches and considerations for monitoring.

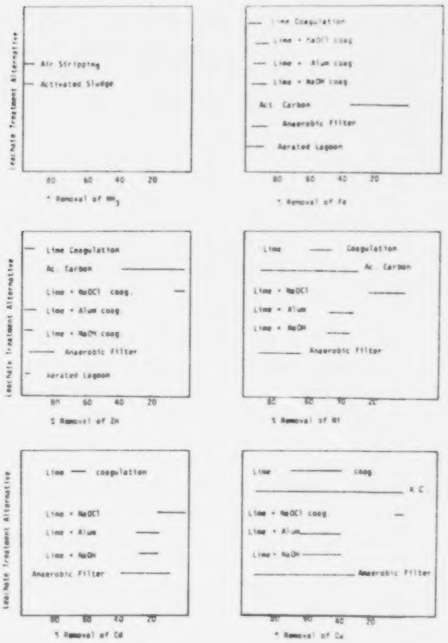

Figure 10. % removal of NH_3, Fe, Zn, Ni, Cd, and Cu by various treatment alternatives

The first two sections address available monitoring methods, costs, and the advantages and limitations of the monitoring equipment for these two hydrologic systems. The final section discusses the selection of sampling areas, monitoring frequency, monitoring parameters, and sample collection and preservation technology. Detailed information is provided in Lu et al. (11).

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LEACHATE PRODUCTION BY LANDFILLED PROCESSED MUNICIPAL WASTES

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ABSTRACT

The purpose of the project was to evaluate waste processing methods used prior to landfilling solid wastes by comparing the leachates produced by wastes that were processed in different ways. Five lysimeters (buried concrete landfill simulators) were loaded with municipal solid wastes that were: (1) shredded and baled, (2) baled, (3) baled and saturated with water, (4) shredded, and (5) unprocessed, respectively. Leachate samples were collected to determine moisture balances and leachate pollutant concentrations in order to evaluate the processing methods. After equal volumes of water were applied to all five lysimeters, the baled wastes were found to produce large quantities of dilute leachate, while the shredded waste produced smaller quantities of concentrated leachate. Gas composition data from the lysimeters was inconsistent but did not indicate any obvious differences among the waste processing methods.

Introduction

The objective of this project was to study the leachate produced by landfilled municipal solid waste. Five lysimeters (simulated landfills) were used to compare the leachate produced by refuse which was processed in five different ways before placement into the lysimeter cells. The operating conditions are the same for all five lysimeters (except moisture saturation for Cell 3) so that if the emplaced wastes are assumed to be identical. any identical, any differences in leachate emanating from the lysimeters should be due to differences in processing of the refuse prior to placement in the cells.

The processing procedures of the refuse in the lysimeters are as follows:

Cell 1	Baled, shredded refuse
Cell 2	Baled, unshredded refuse
Cell 3	Baled, unshredded refuse saturated with water
Cell 4	Shredded refuse
Cell 5	Unshredded, unbaled refuse (control)

This report will compare the refuse processing methods by observing their effects on moisture balances, leachate composition, and gas compositions for the lysimeters. The cells have been monitored continuously since January 1975, and monthly data taken through May 1980 is presented and used for graphical representations and predicting of trends.

Lysimeter Operation

The five landfill cells are located in Franklin, Ohio. They are buried concrete cells containing municipal refuse which had been preprocessed in different ways. The refuse was placed in the lysimeters as described in the following table.

Cell	Refuse	Wet Weight
1	Baled, shredded	11,700 kg
2	Baled, whole	11,700 kg
3	Baled, whole, saturated with water	11,700 kg
4	Unbaled, shredded	10,800 kg
5	Unbaled, whole	9,700 kg

The refuse wet weights for Cells 1 through 3 were calculated using average bale weights. The cells were instrumented and plumbed for temperature monitoring, gas sampling, leachate collection, and water addition.

Water additions were made monthly to Cells 1 through 5 according to a schedule designed to reflect the rainfall patterns of the midwestern region of the United States. The schedule is presented in Table 1.

Leachate collection are performed via plumbing connecting each lysimeter to a central instrumentation cell. Leachate samples are taken monthly and analyzed for composition. Leachate is also quantitatively drained from the cells each month for water balance analyses.

Moisture Balance

Landfill Cells 1, 2, 4, and 5 have been operated on a monthly water addition-volumetric leachate collection basis since January 1975. Water additions are made each month according to the schedule, and all leachate is drained each month also.

Cell 3 has been operated during this time on a saturation basis. In the initial stages of the project, only the minimum volume of leachate required for analyses was withdrawn each month. After the refuse became saturated with water, a volume of leachate equal to the volume of water added the next month was removed each month. Since no evaporation or transpiration occurs within the sealed cells, the moisture content of Cell 3 is assumed to remain constant.

When refuse is landfilled, it will absorb moisture which percolates through the soil cover. The refuse will continue to absorb water until its field capacity is reached. At this point, the amount

TABLE 1. SCHEDULE OF WATER ADDITION

	Water to be applied		
Month of year	Centimeters equivalent water	gallons	liters
January	0	0	0
February	5.54	104.5	396
March	11.08	209	791
April	11.08	209	791
May	11.08	209	791
June	5.54	104.5	396
July	0	0	0
August	0	0	0
September	0	0	0
October	5.54	104.5	396
November	5.54	104.5	396
December	5.54	104.5	396

of water infiltrating the landfill should equal the amount of leachate coming from it. But this is not exactly what happens in an actual landfill situation. Fenn's definition of apparent or practical field capacity more closely describes what actually occurs in nature. Fenn(2) defines field capacity as the maximum moisture content which a soil (or refuse) can retain in a gravitational field without producing continuous downward percolation. In this project, "field capacity" is assumed to have been reached when continuous leachate production has occurred.

The total moisture content of a cell can be calculated by subtracting the volume of leachate removed from the volume of water added. Since these cells are enclosed, evaporation, transpiration, and runoff do not occur. The apparent moisture retained in the cell is calculated as follows:

$$M = \frac{1000\ (R - L)}{G}$$

where

M = apparent moisture retained (mℓ/kg),
R = water added (liters),
L = leachate removed (liters),
G = kg dry solid waste.

A summary of the moisture balance data for Cells 1, 2, 4, and 5 is presented in Table 2. These data indicate that the waste processing procedures had an effect on the start of continuous leachate production. The cells containing wastes that were processed (1, 2, and 4) started producing leachate before Cell 5 (untreated waste). Cells 1, 2, and 4 began producing leachate in the fifth, second, and the fifth, second, and fourth months, respectively; while Cell 5, the unbaled whole refuse, only began producing leachate in the sixth month.

It is also shown that Cells 4 and 5, containing unbaled refuse, have retained substantially more moisture than Cells 1 and 2, containing baled refuse. The

TABLE 2. MOISTURE BALANCE SUMMARY

	Moisture retention data (mℓ/kg refuse)			
Date	Cell 1 (5/75*)	Cell 2 (2/75*)	Cell 4 (4/75*)	Cell 5 (6/75*)
12/75	306	316	475	579
12/76	406	426	672	797
12/77	520	548	814	968
12/78	737	774	1,131	1,299
12/79	854	851	1,198	1,309
5/80	906	911	1,225	1,316

* Date continuous leachate production began.

moisture retained values for Cells 1 and 2 are very similar in magnitude. The unprocessed refuse of Cell 5 has retained more moisture than that of Cell 4 (shredded waste). The two cells containing unshredded waste (2 and 5) have retained more moisture than their counterparts containing shredded waste (1 and 4).

The results indicate that baling of refuse has an inhibitory effect on its moisture retention capability. The tightly-packed bales are more difficult for water to infiltrate than loose refuse, and so absorption of water proceeds at a slower rate. Edens(1) also found that baled waste had absorbed little moisture 1 year after emplacement and after 8 months of leachate outflow. Core samples of the baled waste contained only 1 percent more moisture than what was originally present.

Shredding of refuse is also shown here to decrease moisture retention, but to a much lesser degree than baling. Fungaroli(3) reported that milling (shredding) of refuse increases greatly the saturated field capacity, but our shredded waste appears to have not reached field capacity yet. When it does reach field capacity, it may well retain more moisture than the unprocessed waste. Fungaroli also noted that increasing the density of milled refuse (by making smaller particles) increases the amount of that refuse remaining at a moisture content less than field capacity. This results in dry parcels within the refuse mass, and this may have occurred to some extent in our shredded refuse.

The moisture balance figures also show that the rate of moisture absorption by the wastes in Cells 4 and 5 has slowed considerably compared to the baled wastes in Cells 1 and 2. Cell 5 retains only a little more moisture at May 1980 than it did at December 1978, and Cell 4's rate of moisture uptake is only slightly more than Cell 5's in the past year. This indicates that the nonbaled wastes of Cells 4 and 5 are nearing their field capacities with Cell 5 perhaps having already reached its field capacity.

Fungaroli found that the field capacity of compacted nonbaled refuse in his lysimeter was 1380 mℓ/kg dry refuse. This number is somewhat larger than our values for nonbaled refuse, and it suggests that our nonbaled refuse is near field capacity. Other field capacity values for nonbaled refuse have been reported as 350, 358, and 400 mm/m refuse depth, which convert to 1,150; 1,140; and 1,260 mℓ/kg, respectively (Wigh[6]).

Leachate Data

Leachate is collected and drained from the lysimeters each month through drain lines at the bottom of each cell. The volume of leachate is measured, and a sample from each cell is analyzed for biological and chemical parameters. The parameters measured each month are acidity, alkalinity, pH, conductivity, oxidation reduction potential, total solids, total volatile solids, total organic carbon, total kjeldahl nitrogen, chemical oxygen demand, sulfide, total phosphorus, and the metals iron, copper, cadmium, zinc, nickel, chromium, and lead. Every third month the leachates are analyzed for volatile acids, biochemical oxygen demand, and chloride.

Because the wastes in each of the five lysimeters were mechanically processed differently before "landfilling," the cells differ in water retention ability. The cells have not produced equal volumes of leachate even though equal volumes of water are added to them each month. The greatest amounts of leachate have emanated from Cells 1, 2, and 3 containing baled waste. Much smaller leachate volumes have drained from Cell 4 (shredded waste), and the least volume of leachate has emanated from Cell 5 (unprocessed waste). But the volume of leachate produced alone is not indicative of its pollution potential. The leachates from Cells 1 and 2 contain less pollutants per liter than the leachate from Cell 4. The concentrations of the measured pollutants in the leachate and also the cumulative masses of pollutants in the leachate are found to be the least in Cells 1 and 2 and the greatest in Cell 4.

The differences in waste composition among the cells and the differences in leachate volumes produced necessitate the use of some basis to compare leachate pollutants. The measure of leachate quality chosen for our purposes is the total mass of specific pollutant emanating from each cell per kilogram of dry solid waste. This is calculated by multiplying the measured concentration of pollutant times the volume of leachate, divided by the total mass of dry solid waste placed in the test cell.

Since the cells do not produce equal volumes of leachate in the same month, the cumulative masses of pollutants from each cell can be compared on a basis of leachate volume or on a basis of time. In comparing mass removals of a pollutant on a leachate volume basis, the cumulative mass removed by a standard volume of leachate is compiled for each lysimeter. The same volume of leachate is used for all the cells, so the cell producing the most concentrated leachate will have the largest mass removal. Comparing mass removals on a time basis does not take into account the volumes of leachate emanating from the cells; it only considers the total mass of pollutant removed in a given time period. More pollutants may be removed in a small volume of concentrated leachate than in a large volume of dilute leachate.

Table 3 contains the cumulative masses of leachate parameters removed from the lysimeters on an equal leachate volume basis. The numbers shown are the total masses of parameters removed in 12,401 liters of leachate. This is the total volume of leachate which has thus far emanated from Cell 5. It is seen that Cell 4 (shredded waste) has the greatest mass removals of 4 out of the 5 biological parameters (alkalinity, total solids, TOC, TKN, and COD), and has the second largest mass removal of TKN. This finding was expected since shredding of refuse makes infiltration of the water more rapid, accelerating the decomposition process. Conversely, baling of refuse inhibits water infiltration and produces a more dilute leachate. Cells 1 and 2, containing baled refuse, have lower mass removals of the biological parameters than the

TABLE 3. CUMULATIVE MASSES OF LEACHATE PARAMETERS PER KG DRY REFUSE ON A BASIS OF EQUAL LEACHATE VOLUME

Parameter	Cell 1	Cell 2	Cell 3	Cell 4	Cell 5
Alkalinity, mg/kg	5,450	6,960	9,420	17,200	9,280
Total solids, mg/kg	10,500	15,200	21,200	40,200	24,200
TOC, mg/kg	6,040	9,950	13,300	21,700	16,500
TKN, mg/kg	241	411	514	712	729
COD, mg/kg	9,230	23,200	36,000	50,000	34,600
Iron, mg/kg	273	714	1,180	1,940	540
Copper, µg/kg	149	152	490	152	128
Cadmium, µg/kg	60	76	78	112	84
Zinc, mg/kg	13	20	10	200	42
Nickel, µg/kg	539	789	1,060	1,950	759
Chromium, µg/kg	246	304	305	284	167
Lead, µg/kg	598	654	564	856	577

other three cells. Cell 3 (baled, saturated) has mass removals roughly equivalent to those of Cell 5 (unprocessed) for the biological parameters. It may be that the saturation of the waste in Cell 3 has overcome the inhibition of moisture infiltration normally produced by baling, thus causing more leaching of pollutants than would usually be expected from baled refuse. The mass removal values for Cells 3 and 5 are generally in in between the values for Cells 1 and 4 and Cells 2 and 4.

Before comparing the cumulative masses of metals leached from the cells in Table 3, it should be noted that most of these metals have been detected in only trace amounts in the monthly leachate samples. This means that any analytical error in the detection of these metals would have a greater effect on the calculated cumulative mass leached than an error in the analysis for a leachate parameter which has a high concentration, such as TOC or COD. Any variations in the metals composition of the wastes among cells would then cause a large effect in the composition of the leachate. In many instances, the concentration of a metal in the leachate was below the detection limit of the method. Rather than taking the concentration as zero mg/ℓ, the detection limit itself was taken as the concentration, and the total mass of metal leached was calculated from that value. This practice gives us an artificially high number for the cumulative mass removal, but it is better to do this than to assume that no metal is leached by a large volume of leachate, simply because the metal cannot be detected.

Just as Cell 4 had the greatest mass removals for the biological parameters, it has the greatest mass removals of five of the seven metals. The high relative numbers for iron and zinc especially point out the enhanced rate of leaching caused by shredding of the waste. In general, the metals results do not show the inhibition of leaching by baling of the waste since Cells 1 and 2 have mass removals approximately the same as those of Cell 5 (control). Only the mass removals of zinc from Cells 1 and 2 are clearly indicative of a reduced rate of leaching when compared to the cumulative zinc emanated from Cell 5. The great deal of variability in the metals results is probably caused by the large error possibilities associated with the detection of very low concentrations of copper, cadmium, nickel, chromium, and lead in the leachates. Since the amounts of these metals are low in the waste placed in the lysimeters, much variation may be present from cell to cell in the quantity of metal originally available for leaching. This makes comparisons among the cells for these parameters difficult if not impossible.

The plots of cumulative masses of biological parameters leached vs. cumulative leachate volume (Figures 1 through 5) further indicate the effects of the waste processing procedures. In every case, the shredded refuse (Cell 4) has had more mass leached than the waste processed by other means. The shredding has increased the surface area of the waste and enhanced moisture infiltration, providing more sites for biological and chemical activity. The baled wastes of Cells 1 and 2 have had less cumulative masses removed than the control (Cell 5) showing the inhibition of moisture infiltration and leaching caused by baling of the waste.

Looking again at the cumulative masses of pollutants leached from the cells on an equal leachate volume basis (Table 3), it is interesting to compare the data from Cells 1 and 2. Both cells contain baled waste, but the waste in Cell 1 was shredded before baling. A greater mass of every pollutant has leached from Cell 2 than from Cell 1. More than twice as much COD has leached from Cell 2 than from Cell 1. It appears that baling of shredded waste produces bales that are even more resistant to leaching than

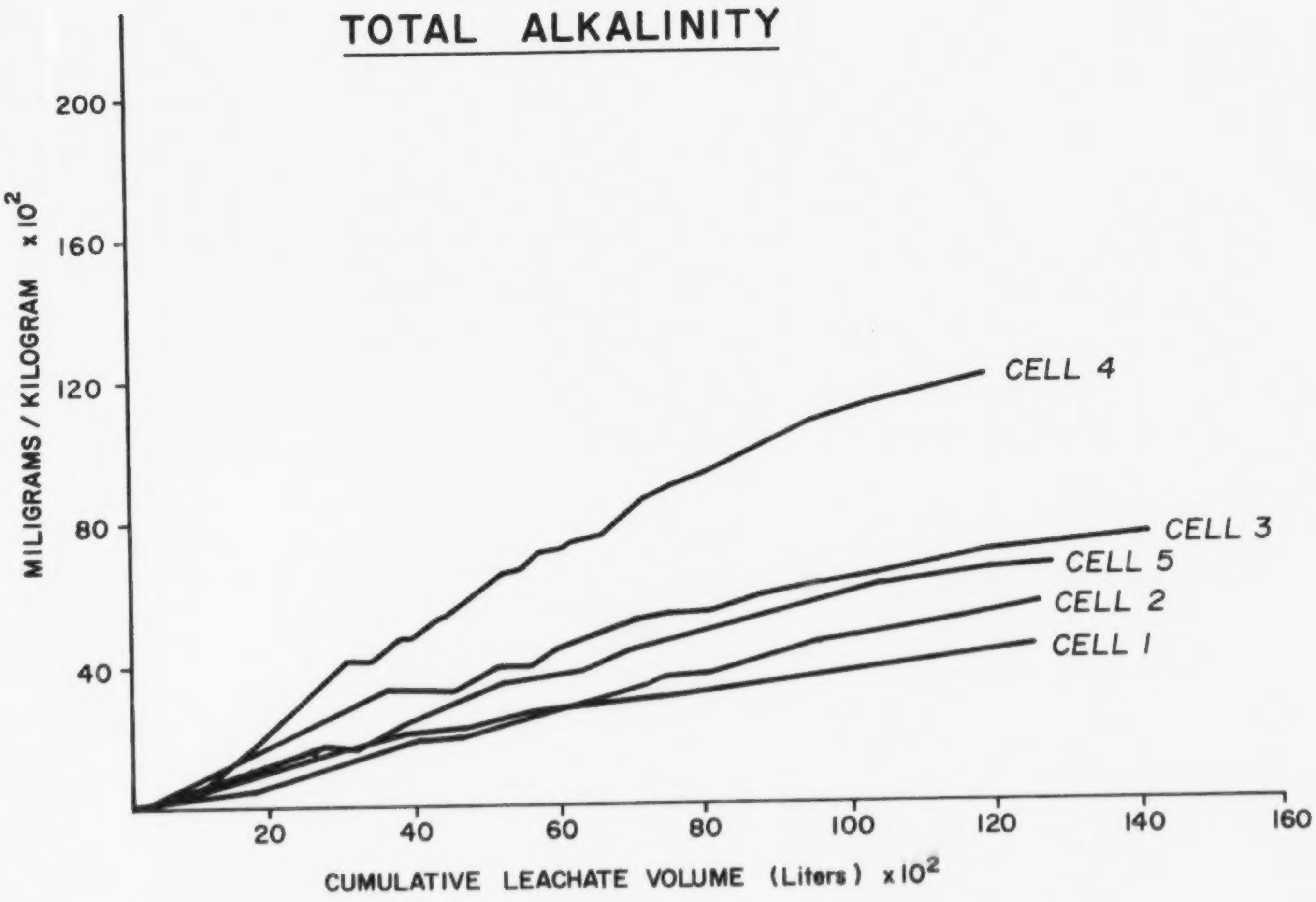

Figure 1. Total alkalinity - mass flow vs. cumulative leachate volume.

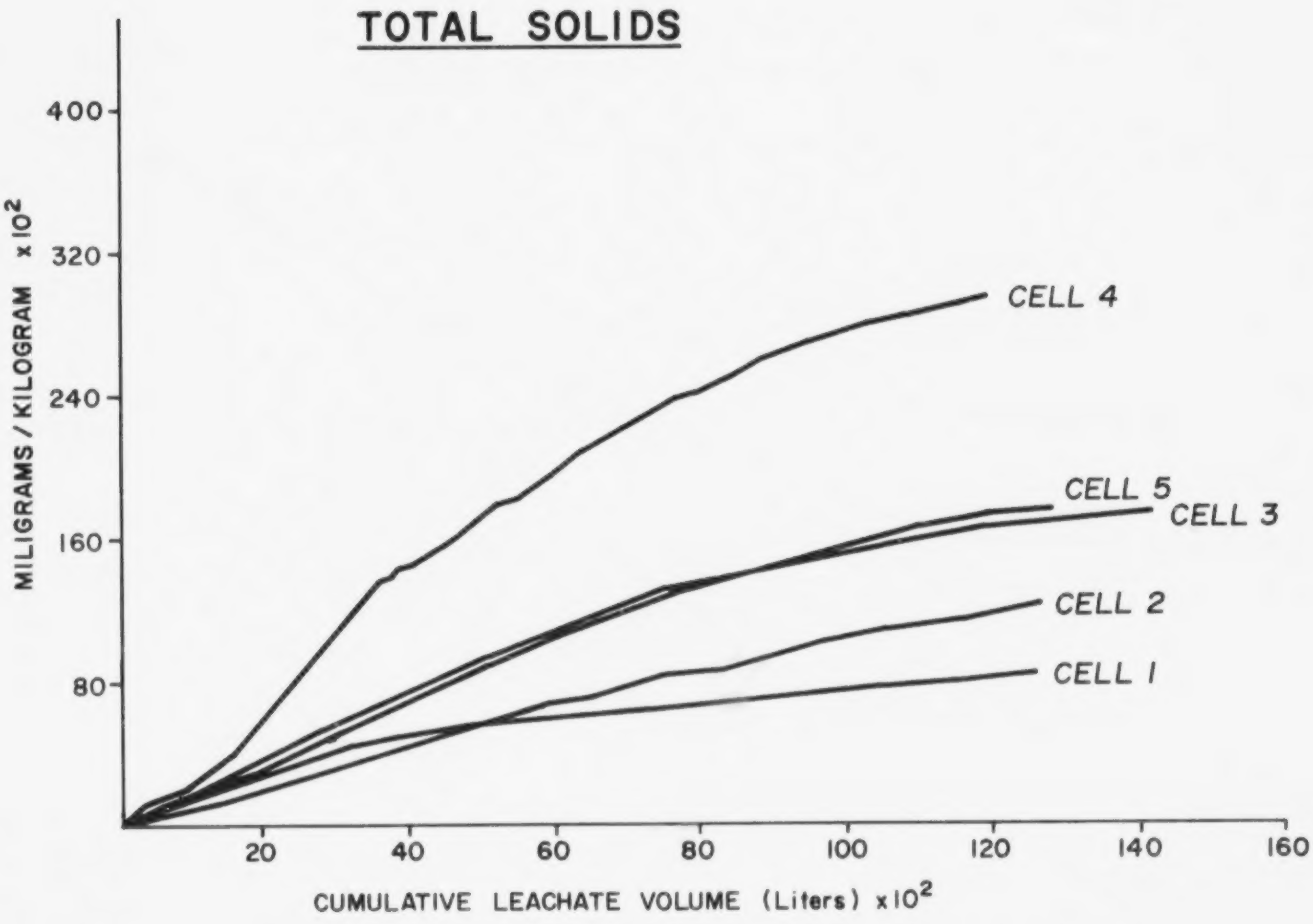

Figure 2. Total solids - mass flow vs. cumulative leachate volume.

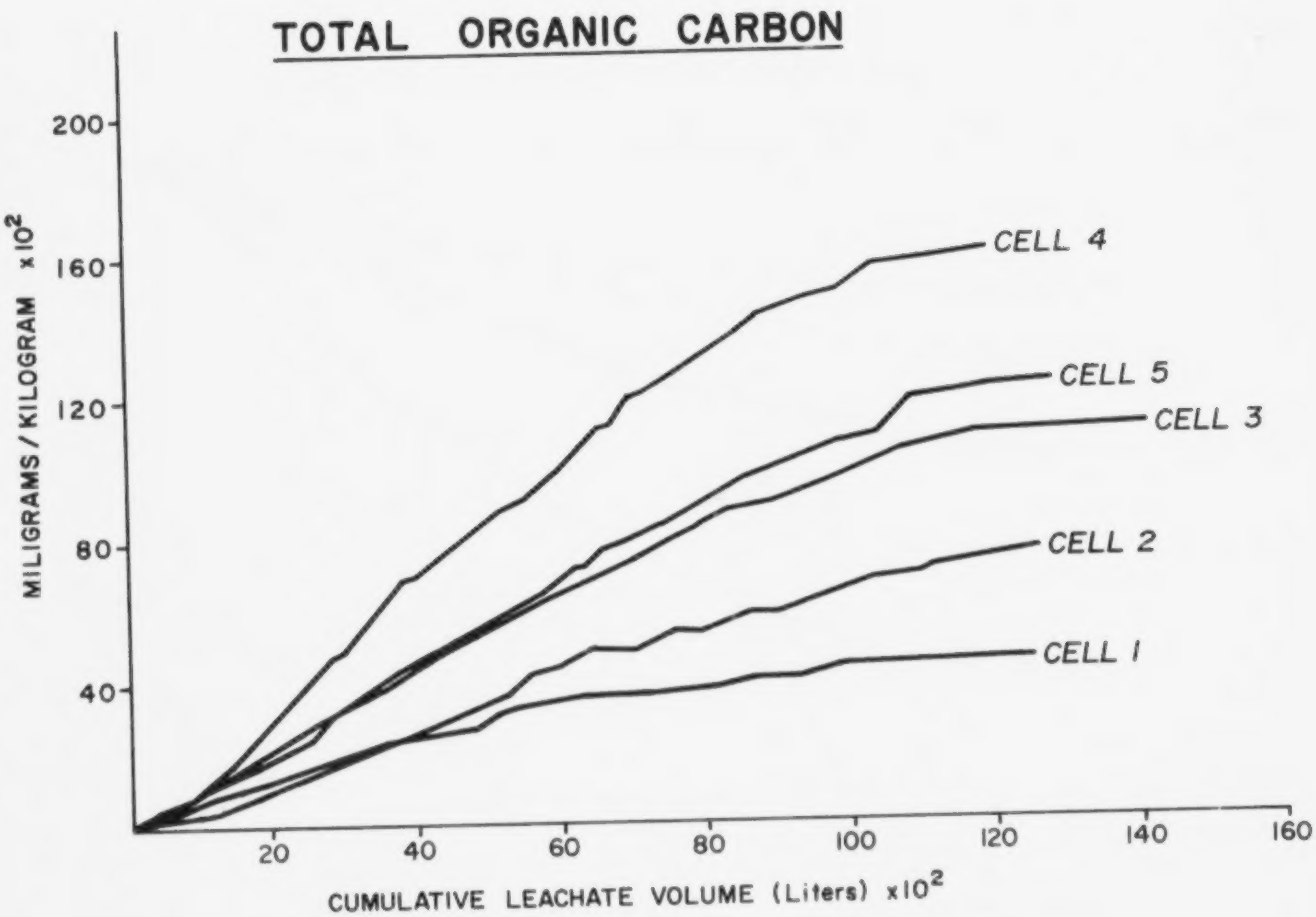

Figure 3. Total organic carbon - mass flow vs. cumulative leachate volume.

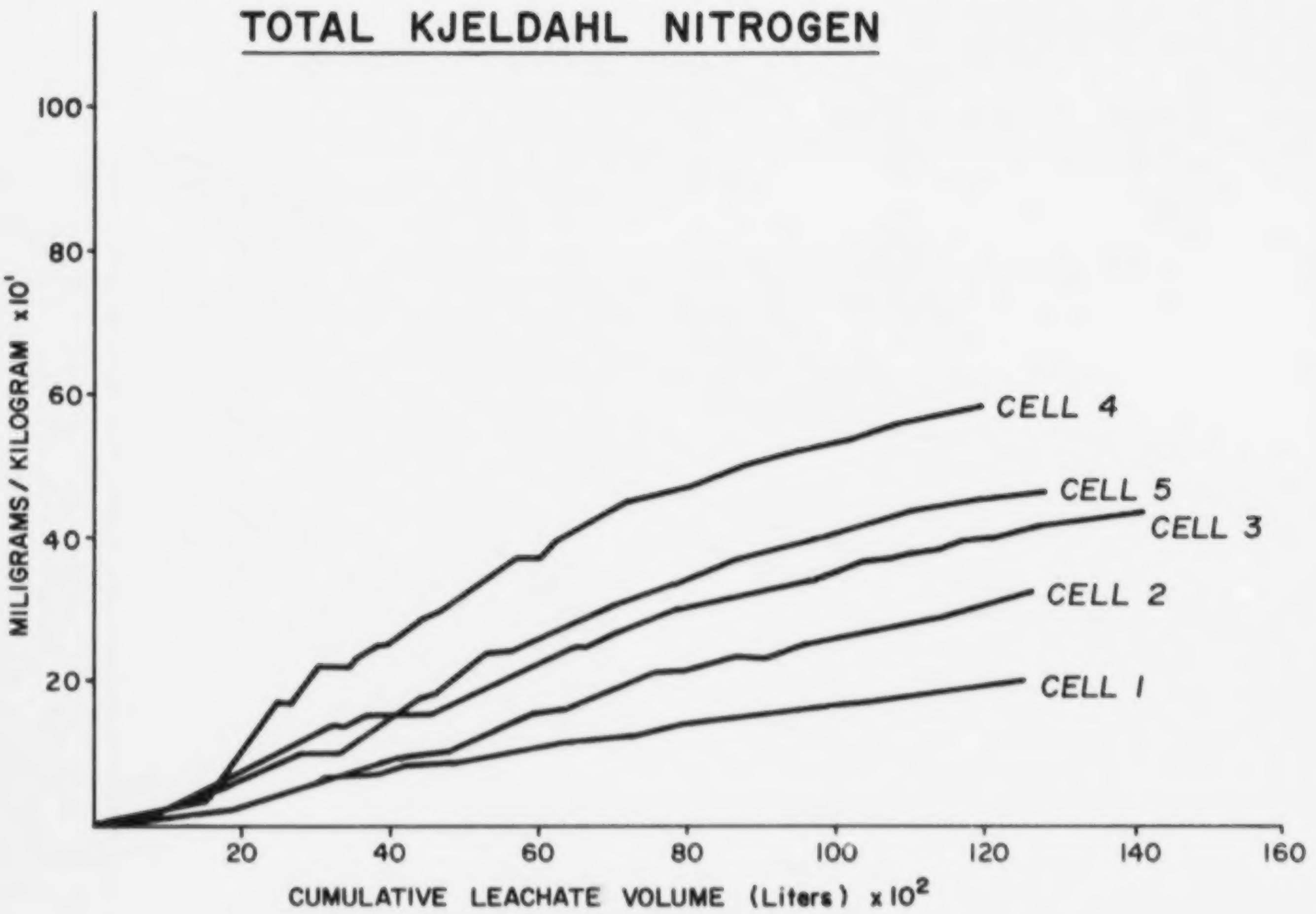

Figure 4. Total kjeldahl nitrogen - mass flow vs. cumulative leachate volume.

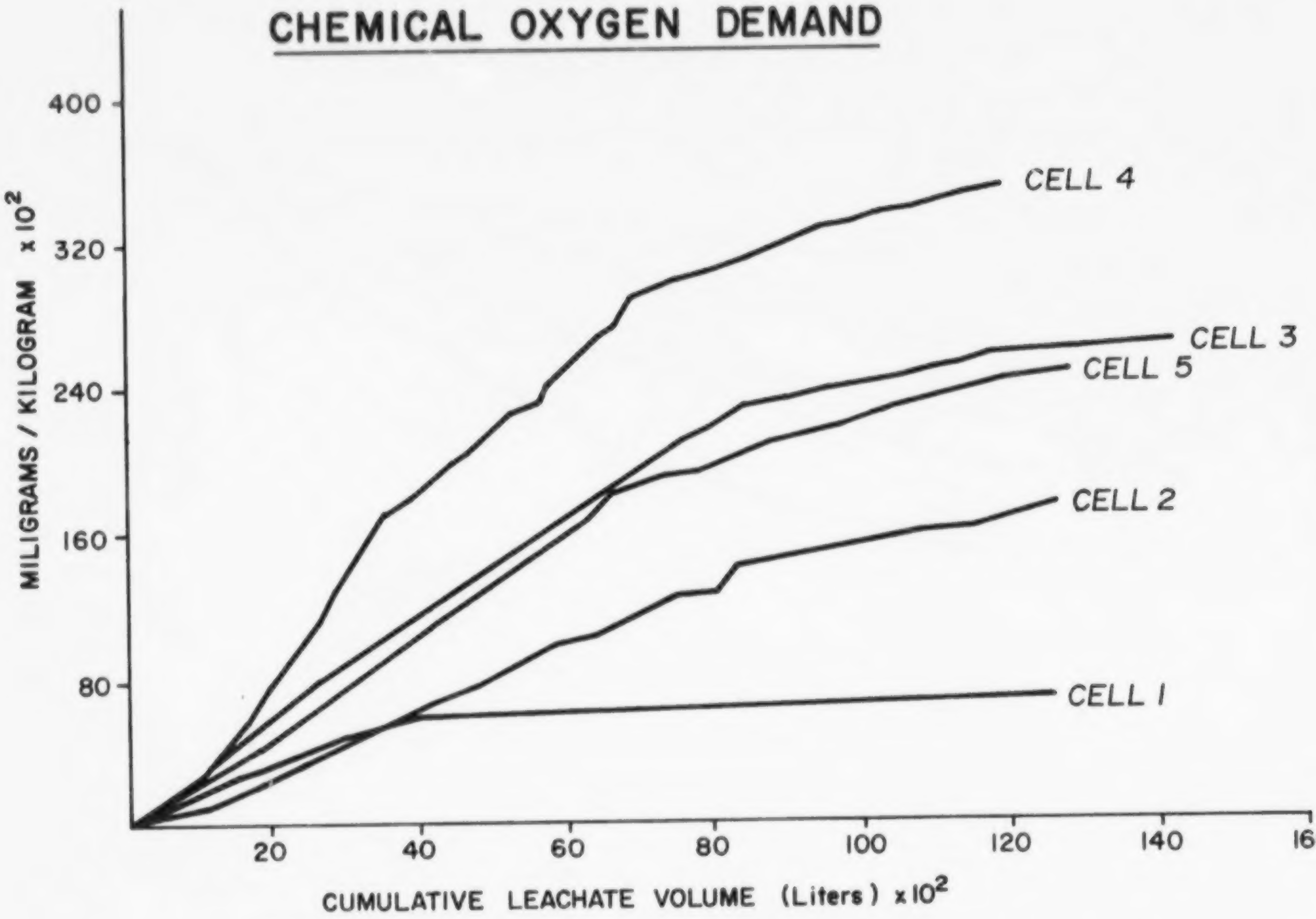

Figure 5. Chemical oxygen demand - mass flow vs. cumulative leachate volume.

bales of nonshredded waste. Reducing the size of refuse particles allows them to "fit together" more compactly in bales, reducing air space between particles more than simple baling. This process would further retard movement of water in and solids out, resulting in a lower rate of leaching for baled shredded waste than for baled waste. The increased rate of leaching of Cell 2 over Cell 1 is also readily evident in the plots of cumulative masses leached versus cumulative leachate volume.

One of the metal parameters deserves special mention in our discussion because of its high rate of leaching from Cell 4 as compared to the other lysimeters. The cumulative mass of zinc leached from Cell 4 is about one order of magnitude higher than that of the other cells. The presence of zinc alkaline batteries in the waste may be responsible for this phenomenon, according to Hentrich.(5) The batteries contain zinc salts which would normally be sealed within, unless the battery were shredded. The zinc salts would then be released and available for leaching. Such batteries would also be expected to be shredded in the waste of Cell 1, but the baling of the waste would hinder leaching of the salts by making the waste more impermeable to water. Rapid leaching of zinc in Cell 1 has not been observed. The disproportionately large significance of this one component of the waste stream points out how minor variations in the compositions of wastes can have great impacts on the character of the leachates emanating from them. However, these variations only assume great importance when the parameter being considered is present in a very small quantity in the refuse.

Gas Composition Data

The biological decomposition of municipal solid waste occurs in the lysimeters in the same manner as in conventional landfills. This transformation of organic materials takes place in three stages, according to Ham.(4) The first stage is aerobic decomposition in which organic matter is degraded to CO_2, water, and partially degraded residual organics. This process is rapid and produces heat. The CO_2 produced dissolves into any water present, decreasing the pH.

The second stage is decomposition by facultative acid forming bacteria. Carbon dioxide is produced rapidly with little heat production. Partially degraded organics, especially organic acids, are formed. The pH is lower than in the aerobic stage, which helps dissolve inorganics.

The third stage is decomposition by anaerobic methane forming bacteria. This process occurs slowly and efficiently, producing CO_2, water, and methane with little heat. Because the decomposition rate in this stage is slow and the waste is becoming more stable in nature, the CO_2 content of the gas decreases and the moisture pH increases, reducing the amount of dissolved inorganic material. This stage is called the methanogenic stage because methane is produced.

These three stages of biological decomposition in landfills are not exclusive of each other. Normally, two or three of them occur simultaneously in different locations within the landfill. The methane formers are symbiotic with the acid formers, practically assuring that some of the organisms present in the facultative stage are also present in the methanogenic stage.

Figures 6 through 10 show time histories of the compositions of gases collected from the test cells. The graphs indicate inconsistencies both in time and from cell to cell. An anaerobic, methane producing situation is indicated for Cell 1, while Cell 5 is shown to have produced little methane. Cells 2 through 4 have produced methane in varying amounts.

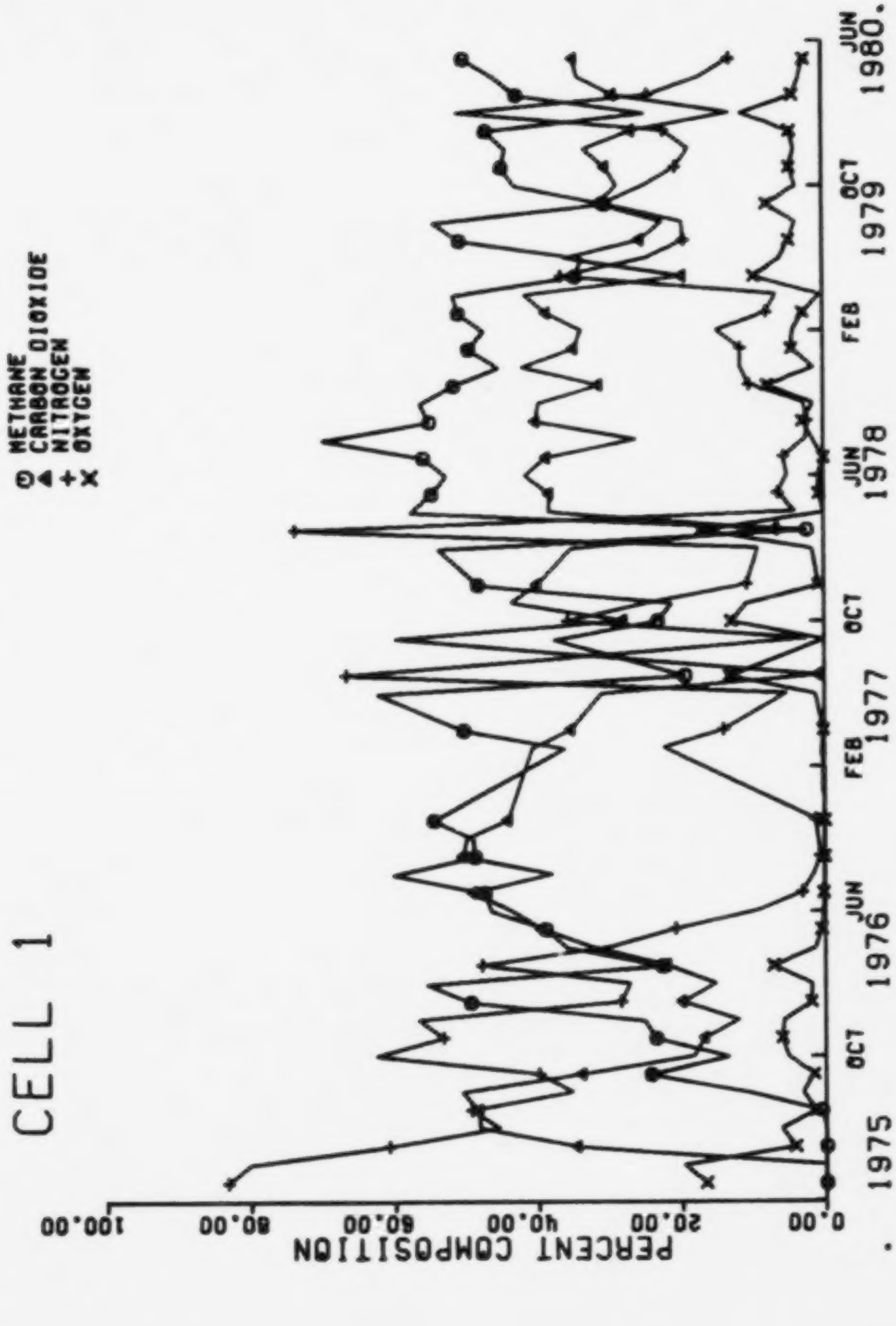

Figure 6. Cell 1 - Gas composition vs. time.

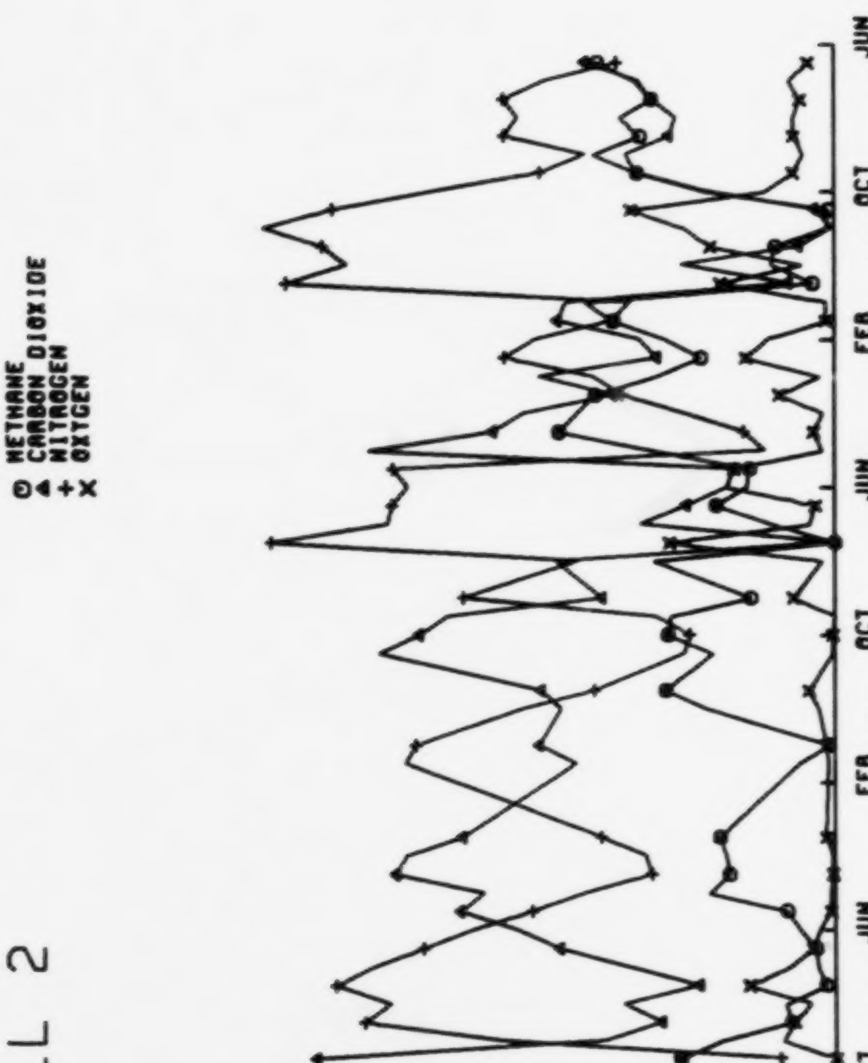

Figure 7. Cell 2 - Gas composition vs. time.

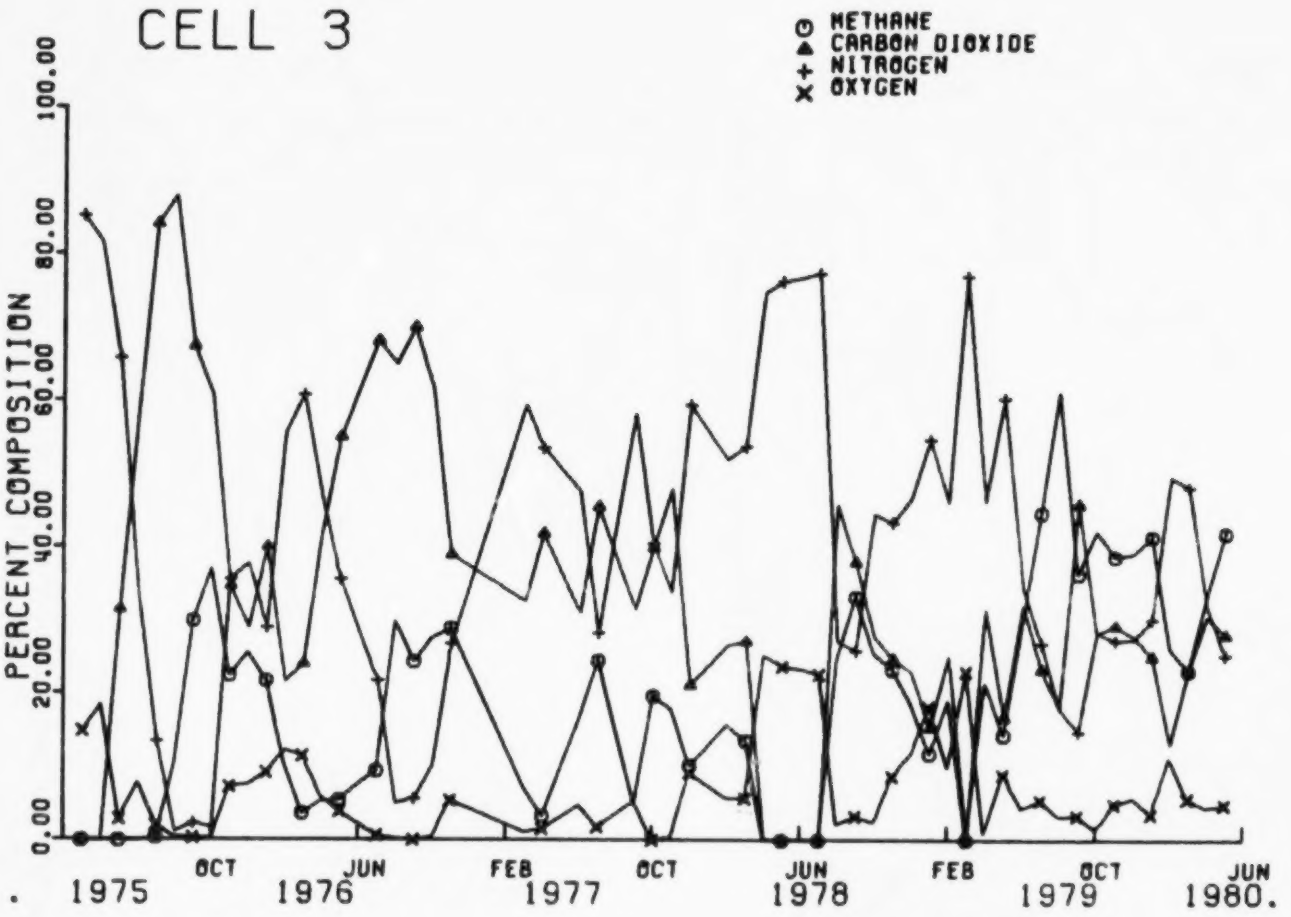

Figure 8. Cell 3 - Gas composition vs. time.

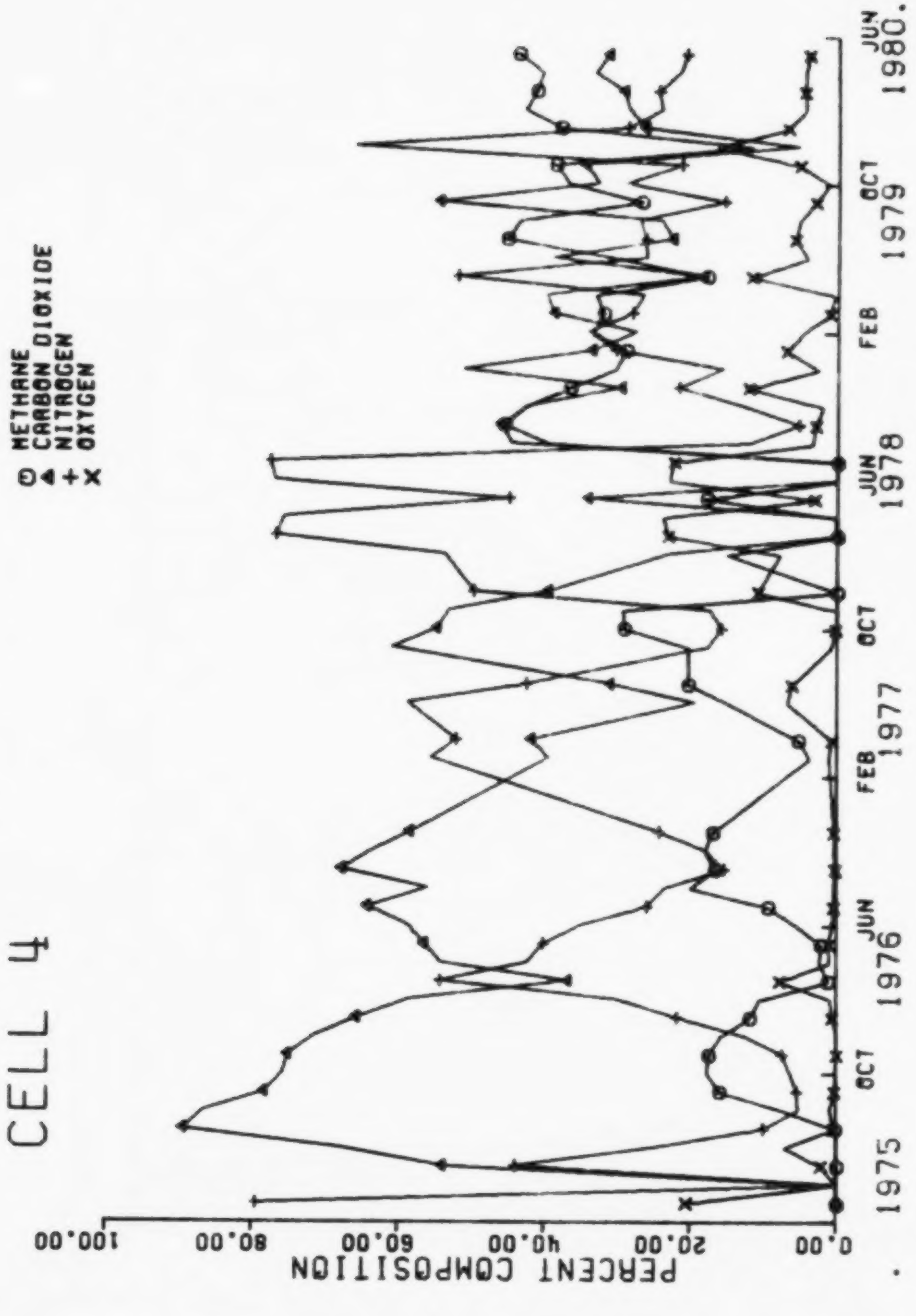

Figure 9. Cell 4 - Gas composition vs. time.

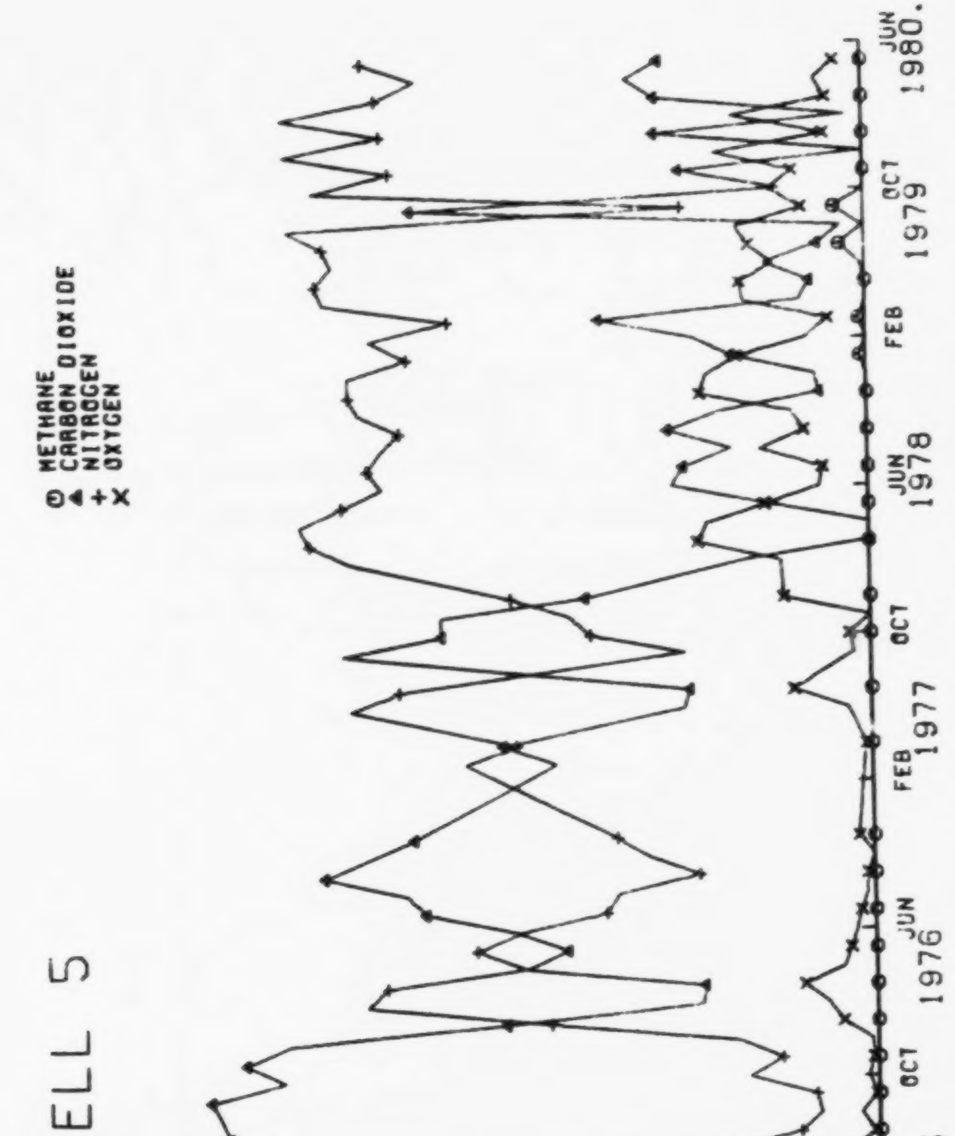

Figure 10. Cell 5 - Gas compostion vs. time.

It seemed unbelievable that no portion of the large mass of waste in Cell 5 had become anaerobic and produced methane. The possibilities of blockages or leaks in the gas collection system of Cell 5 were investigated, and the gas collection probes within the cell were found to have been broken off by settlement of the refuse. Where the gas collection lines went through the walls of the cell, they now met settled clay and gravel which was placed originally over the refuse. New gas collection probes were installed further below the old ones, into the refuse mass. Samples taken after this were as much anaerobic as those from the other cells (~5 percent O_2), but still only traces of methane were detected. The pH of the leachate from Cell 5 has remained below 6, while the leachates from the other cells are all above 6. This leads us to believe that the refuse in Cell 5 is being maintained in a state of acid fermentation, with only few methane forming bacteria present. This would explain the lack of methane production, but just why Cell 5 alone is acting this way is unknown.

In general, the gas composition data do not indicate any obvious effects of the different refuse processing procedures. It is still unknown what effects baling and shredding have on landfill gas production.

Discussion of Results

The analysis of the data collected in the course of the project from the beginning until now gives rise to the following conclusions.

The processing step of baling has been shown to enhance leachate production both by decreasing the time period before continuous production of leachate and by increasing the overall volume of leachate produced. The cells containing baled waste produced leachate about 2 months before the cells containing unbaled waste and also produced more leachate over the entire period of the study. Baling reduces the moisture-retention ability of the waste, as does shredding; but shredding enhances moisture infiltration at the same time. Baling has a much greater positive effect than shredding on the volume of leachate produced.

Baling was seen to result in the formation of leachate that contains relatively low concentrations of pollutants, while shredding of the waste causes leachate that has high concentrations of pollutants. A comparison of the mass flows of pollutants from the cells on a basis of equal leachate volume shows the decreasing effect of baling and increasing effect of shredding on pollutant concentrations in the leachate.

Taking together the leachate volumes and pollutant concentrations resulting from baled vs. shredded waste, it is evident that the two processes produce opposite results. Baling of waste results in large volumes of dilute leachate emanating from a landfill with a long period of time required for stabilization of the landfill. Conversely, shredding of waste results in the formation of smaller volumes of highly concentrated leachate, and a shorter period of time passes before landfill stabilization.

The leachate data from the baled waste in a saturated environment is coincidentally very similar to the data from the waste in the control lysimeter. It may be that these two factors, baled waste and saturated environment, have mutually canceling effects when present in combination. This finding might be of interest in designing landfills in areas where the water table is high.

The data also show that shredding of waste before baling inhibits leaching even more than baling alone. If the slowest possible rate of leaching and longest time period before landfill stabilization were desired, shredding and baling both could be employed.

The only conclusion that can be drawn from the gas composition data is that there is a great deal of variability both among the cells and over time in each cell.

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Recovery of Fecal-Indicator and Pathogenic Microbes from Landfill Leachate

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ABSTRACT

With ever-increasing amounts of municipal and commercial solid waste (garbage, sewage sludge, hospital waste, etc.) entering sanitary landfills and open dumps, the potential exists for human and animal pathogens in these wastes being leached out by rainwater to contaminate ground and surface waters. To determine if this leaching water (i.e. leachate) could contain human pathogens, leachates from commercial and large scale experimental landfills and laboratory lysimeters containing various solid wastes were studied.

Microbial assays on active landfills demonstrated that the leachate contained high concentrations of Gram-negative rods, some of which were identified as *Enterobacter aerogenes*, *Citrobacter* sp., *Salmonella* sp., *Serratia marcescens*, *Proteus* sp. and *Aeromonas* sp. The Gram-positive bacteria present were *Clostridium perfringes*, *Streptococcus faecalis*, *S. faecium*, *S. viridans*, and *Mycobacterium* sp. All of these bacteria may be present in human feces, and several are pathogenic for humans. Landfills inactive for 2 to 6 years had very low concentrations of Gram-negative rods. The most abundant microorganism present were *Streptococcus faecalis*, *Corynebacterium* sp., *Bacillus* sp., yeasts and molds. Pathogens found in these old landfills were *Listeria monocytogenes*, *Acinetobacter* sp., *Moraxella* sp., and *Allescheria boydii*. Newly constructed laboratory lysimeters containing a variety of municipal wastes, demonstrated very low coliform levels in the leachate after 4 months. The streptococcal levels remained higher, but eventually dropped below the detection level after 1½ years. However, the laboratory lysimeters containing sewage sludge continued to discharge fecal microorganisms, both coliform and streptococci, even after two years. Fecal-indicator bacteria were isolated in the solid waste from lysimeters 2 years old, and from a 9 year old fullscale landfill . These results demonstrate that fecal-indicator bacteria and some pathogenic forms do not die as rapidly in landfills as previously reported but survive for extended periods of time with the potential to contaminate ground and surface waters when released in leachates from the landfills.

INTRODUCTION

This research is concerned with the potential contamination of water supplies by pathogenic microorganisms contained in leachates from sanitary landfills. The ever increasing use in the United States of landfills as disposal sites for garbage; other municipal waste as pet feces, animal remains, and disposable baby diapers containing human infant feces; respiratory waste in paper tissues; hospital wastes; and raw or partially treated sewage sludges poses a potentially significant public health hazard. This is especially true when consideration is given to the possible contamination of adjacent ground and surface waters by microbial migration through the landfill site and the adjacent soil.

Specific information on the survival of infectious agents in sanitary landfills is essentially nonexistent. Also, few studies exist which characterize (i.e. identify) the specific microbial populations occuring in the leachates from landfills. Several investigators, however, have analyzed landfill leachates for total coliforms, fecal coliforms, and fecal streptococci. The survival of such fecal indicator-bacteria in leachate suggests the presence of microbial pathogens in the leachate [See Scarpino (22,23) for a discussion of the microbial indicator concept]. One of the first such studies on landfills was reported by Carpenter and Setter (5) who found a mean of 10^5/100 ml levels of *Escherichia coli*-type coliforms in New York City landfills that were less than two years old. Sobsey (25) found both total coliforms, and fecal coliforms in several inactive landfills, indicating that coliforms remain alive for an indefinite length of time after landfills have been closed.

Specific microorganisms were identified from old landfills by Cook et al. (6). They found *Escherichia coli*, *Serratia marcescens*, *Streptococcus faecalis*, *Bacillus*, *Pseudomanas*, *Achromobacter*, *Flavobacterium*, *Lactobacillus*, *Sarcina*, *Propionibacterium*, and *Clostridium*. They, also, noted the presence of many saprophytic molds such as *Aspergillus*, *Penicillium*, *Fusarium* and others. Other investigators have monitored leachates from a variety of commerical and experimental landfills to determine densities of total coliforms, fecal coliforms and fecal streptococci. For example, Blannon and Peterson (3) looked for fecal coliform and fecal streptococcus levels in leachates from an experimental landfill containing 435 tons (wet weight) of residential solid waste. Biochemical characterization of 60 streptococcal isolates showed that 33.3% were indicative of fecal contamination from warm-blooded animals, and included *S. faecalis* biotypes II and III (12 strains), *S. durans* (3 strains), and *S. equinus* (5 strains).

Thus, the objective of this investigation was to evaluate the pathogenic microbial presence in leachates, considering also the fecal-indicator microbes, especially the fecal streptococci. In this manner we hoped to obtain information as to the health hazard associated with landfill leachates. Specifically this research included:

1. The determination of bacterial and fungal pathogens in leachate from several existing commercial and experimental landfills.
2. The characterization of the species of fecal streptococci persisting in landfill leachates to ascertain their significance relative to the pathogenic presence in the same leachates.
3. The construction of small-scale laboratory experimental lysimeters to study microbial population changes in leachates.

The experimental work described in this report is in two phases, each varying with the landfill sites used. The first phase involved analyses on the active and inactive landfill sites built before 1977. The second involved a study of 6 newly constructed lysimeters, each containing different solid wastes.

MATERIALS AND METHODS

Sampling Sites

A list of the landfill sampling sites and their characterization are given in Table 1. The last site mentioned in this table is further described in Table 2. A full description and construction details of these sites were reported by Scarpino et al. (24). Of all of the sites listed in Table 1, only the commercial landfill received waste daily and was thus active, while the rest were inactive.

Sampling Procedure

Leachate obtained from the commercial landfill was collected at the point where it flowed from the surface of the landfill slope. Occasionally, leachate was obtained from horizontally placed steel

TABLE 1. LANDFILL SAMPLING SITES

Landfill	Leachate or Solid Waste Abbreviation	Dimensions and Placement	Solid Waste Added		Date Prepared	Date of Last Waste Deposit
			Type	Amount		
Commercial, Full-Scale, Operational	CL	700 Acres in Soil	Municipal, Hospital, Commercial	Millions of Tons	1954	Daily (Active)
Boone County, Kentucky	BC 1	40' x 140' x 8.5' in soil	Municipal	435 Tons	1971	June 1971 (Inactive)
Boone County, Kentucky	BC 2 through 5	12' x 6' Steel Cylinders in Soil	Municipal	2 Tons	1972	August, 1972 (Inactive)
Center Hill, Cincinnati, Ohio	CH 1 through 19	12' x 6' Steel Cylinders	Municipal, Sewage Sludge, Industrial	2 Tons	1974, 1975	Nov. 1974 & April, 1975 (Inactive)
Univ. of Cincinnati, Cincinnati, Ohio	A through F	2' x 2' in Steel 55 gallon Drums	Municipal, Sewage Sludge, Hospital	150 lbs	1978	August, 1978 (Inactive)

TABLE 2. DESCRIPTION AND PURPOSE OF LABORATORY LYSIMETERS

Number	Contents*#	Purpose
A	Sewage Sludge	Microbial control for the sewage sludge used in Lysimeter B.
B	Municipal Solid Waste Plus Sewage Sludge	To determine the impact of municipal sewage sludge additions on the rates of decomposition and formation of gas and leachate. Also, to elucidate microbial changes in the leachate.
C	Municipal Solid Waste	Control on microbial numbers and species.
D	Municipal Solid Waste	Same as Lysimeter C.
E	Municipal Solid Waste Plus Hospital Waste	To determine the impact of pathogens on municipal solid waste and pathogen ability to survive landfill conditions.
F	Hospital Waste	Microbial control on the hospital waste used in Lysimeter E.

*208.2 liters in each 55 gallon drum.
#Lysimeters maintained at 20 C in a constant temperature room.

pipes extending deep within the landfill. In all the remaining sites listed in Table 1, the leachate was obtained from sampling ports. All leachate samples were allowed to drain into glass flasks, previously gassed with nitrogen. These flasks were iced immediately, taken to the laboratory, stored at 4 C, and processed in 6 to 24 hours.

Solid waste samples taken from landfills were immediately placed in sterile plastic bags, and these in turn were placed in nitrogen-gassed jars to minimize oxidation of the sample. Those samples to be used for the recovery of anaerobes were placed immediately under anaerobic conditions at the sampling sites using the GasPak (BBL) system. All jars were kept in an ice chest during their transportation to the laboratory. There the samples were kept at 4 C and processed within 24 hours.

Concentration and Digestion of Samples for Mycobacterial Isolation

To isolate mycobacteria from leachate on a consistant basis, it was necessary to both concentrate and digest the samples. The concentration process involved filtering a liter of sample leachate through diatomaceous earth filters. After filtration, the diatomaceous earth was removed, mixed with phosphate buffer, and allowed to settle out. The supernatant liquid was then digested with n-acetyl-L-cysteine and NaOH, followed by centrifugation. This sediment was inoculated onto Lowenstein-Jensen slants and Middlebrook 7H10 Agar, and incubated at 37 C for 2-4 weeks. The digestion procedure destroyed the saprophytes, and allowed the slower growing and resistant mycobacteria to grow.

Sample Homogenization

The solid waste was homogenized in preparation for the enumeration of its bacterial and fungal contents. The solid waste was weighed, diluted 1:10 with phosphate buffer, and minced in a Waring blender for 15 seconds at high speed.

Microbial Enumeration Procedures

The leachate samples were diluted with phosphate buffer and then plated using the spread plate technique, as follows: 0.1 ml of the leachate dilutions were each placed on separate agar plates, and the inoculum was spread over the surface of the agar with a sterile glass rod. Replicate bacterial plates were made, and all were incubated at 37 C for 48 hours and then counted. Fungal agar plates were incubated at ambient temperature for one week and counted.

Blood agar was used to obtain both the aerobic and the anaerobic bacterial counts. The GasPak(BBL) system was used to produce the anaerobic conditions for the anaerobic plates. The total bacterial count was made with brain heart infusion or Standard Methods agar. The differential agars used for the microbial plate counts were: eosin methylene blue (EMB) for the Gram-negative rods; KF-streptococcal for the streptococci; tellurite for nonsporing Gram-positive bacteria; mycosel for dermatophytes; inhibitory mold for systemic fungi; and Sabouraud for yeasts and molds. These procedures are consistent with those found in Microbiological Methods for Monitoring the Environment (4) and Manual of Clinical Microbiology (15).

The most probable number (MPN) tests (1, 11) were used to determine the total coliforms, fecal coliforms, and the fecal streptococcal levels instead of the membrane filter method, as the former method gave higher counts (11). The coliform MPN test procedures were changed during the Phase II experiments from that of Standard Methods (1) to that of Allen et al. (2). Allen et al. had noticed that large numbers of bacteria in environmental waters often interferred with lauryl sulfate broth gas production. The subsequent inoculation of all of the cloudy tubes from the lauryl sulfate broth, without regard to gas production, into brilliant green bile broth and EC medium resulted in significant increases in the numbers of positive tubes, and a resultant increase in the MPN counts.

Enrichment Procedures

Enrichment procedures for the isolation of Salmonella and Shigella were a part of the leachate analysis. The media used for these isolations included tetrathionate, selenite F and GN broths. Five milliliters to 50 ml of leachate were added to 50 ml of the broths. After 24 and 48 hours incubation at 37 C, these media were streaked on XLD or Hoekton enteric agars. The above procedures also conform to those found in Microbiological Methods for Monitoring the Environment (4) and Manual of Clinical Microbiology (15).

Identification of Microorganisms

The identification procedures used for the various microorganisms isolated from landfills were: Gram-negative rods, Wolf et al. (27), Lennette et al. (15), Koneman et al. (12) and MacFaddin (16); for Gram-positive rods, Cowan (7) and Lennette et al. (14); and for fungi Koneman et al. (13) and Lennette et al. (14). On occassions, the API system was used to extend the identification beyond the procedures given above. Also, serological slide tests were used to confirm for suspected Salmonella, Shigella, and Klebsiella.

In addition to the above procedures, extensive identification (i.e. speciation) of the streptococci isolated from solid wastes and leachates were made using two procedures. One (9) was used to place the streptococci tentatively into Lancefield groups A,B,D, etc. These tests included hemolysis, bacitracin sensitivity, hippurate hydrolysis, bile -esculin hydrolysis, and growth in 6.5% NaCl broth. Representative cultures were confirmed with Lancefield grouping using the serological antisera in Streptex tests (Burroughs-Welcome Co., Research Triangle Park, North Carolina.) The second method identified the Group D streptococci as species by the procedure of Pavlova, et al. (18). This latter procedure involved some 15 biochemical characteristics.

Leachate Inhibition Procedure

Leachate inhibition assays were made to determine the degree to which old lysimeter leachate interferred with microbial growth, particularly with respect to Gram-negative rods. Definite concentrations of the test bacteria were added to leachate obtained from the number 19 Center Hill lysimeter. These test solutions were mixed, diluted, and plated on differential agars. Such agars were useful in that they allowed the test bacterial growth while preventing the leachate bacterial growth. The Gram -negative rods were counted after 24 hours at 37 C. and the streptococci were counted after 48 hours. A growth response less than that appearing in the control indicated leachate inhibitory action.

Mycobacterium Antibiotic Sensitivity Tests

Antibiotic sensitivity tests were used to separate suspected isolates into groups for identification purposes. These bacteria were maintained on Middlebrook 7H9 agar. They were then streaked for isolation, transferred to 7H9 broth, and plated on 7H10 agar containing antibiotic discs. Then they were packed into plastic bags and incubated at 37 C for 3 weeks. The resistant growth was recorded.

RESULTS OF PHASE I EXPERIMENTS

Experimental Approach

The initial work on this project was designed to characterize the bacteria and fungi present in commercial and experimental large-scale landfills. As many species of bacteria and fungi were present in the leachate, it was necessary to use a variety of isolation and identification methods. Most bacterial species grew on enriched media such as blood agar and Standards Methods agar. However, differential or selective media allowed only one kind of bacteria to grow and inhibited the rest. Bacteria and fungi measured by these two types of procedures were aerobic and anaerobic bacteria, Enterobacteriaceae, pseudomonads, streptococci, nonsporeforming Gram-positive rods, molds, yeast, total coliforms, fecal coliforms, and fecal streptococci. Isolates were picked from these media and identified.

Microscope Examination of Streptococcus MPN Tubes

Fecal Streptococci MPN counts were first determined on Center Hill landfill leachates, and are reported in Figure 1. To confirm that these results were caused by streptococci, a Gram stain was made on those ethyl violet azide tubes giving a positive reaction, i.e. the appearance of the purple button in the bottom of the tube. The microscopic examination of the stained purple growth showed that these positive results were not always caused by cocci. Indeed, many positive results were caused by Gram-positive rods. Hereafter, all MPN tubes were examined microscopi-

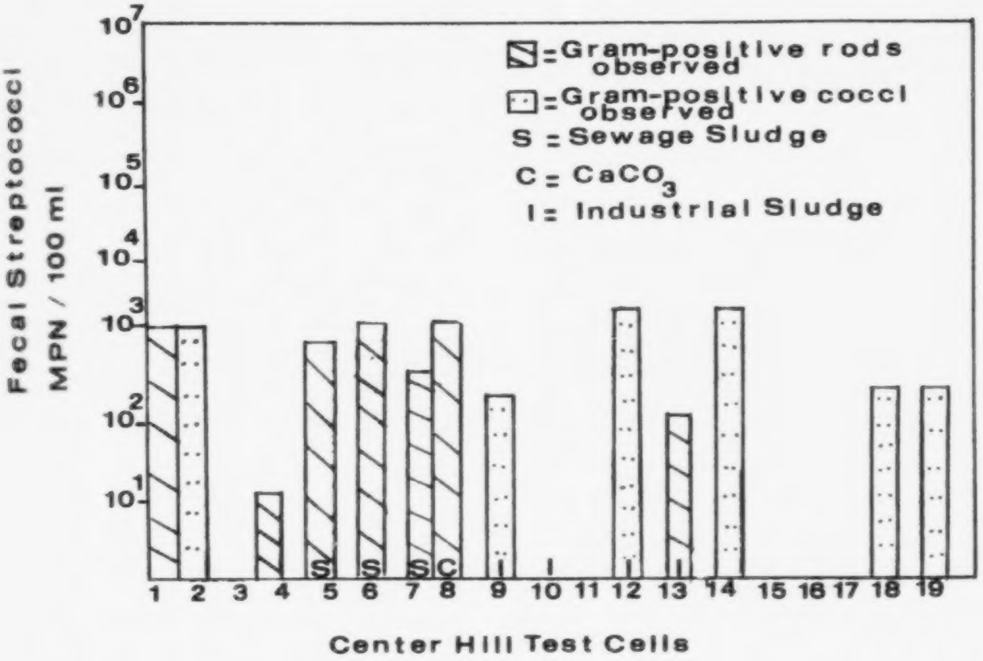

Figure 1. Fecal Streptococcal Presence in Center Hill Experimental Field Lysimeters

cally for Gram-positive cocci, and streptococcal levels were ultimately determined based on coccal presence in positive ethyl violet azide tubes. The significance of these false-positive results is that high counts of streptococci have been reported from these landfills, when this may not always be in fact the case.

In additon, the ethyl violet azide tubes to which 10 ml and 1 ml aliquots of leachate had been added contained fewer or no Gram-positive cocci than those found at higher dilutions. We interpreted this to mean that leachate toxicity was present. It was apparent that as the leachate was diluted out, the toxic agent was also diluted out, because no inhibition of Gram-positive cocci growth was noticed at the higher dilutions. Thus, the toxicity present in the leachate gave false-negative test results and indicated lower concentrations of streptococci than were actually present.

The Enumeration of Leachate Microorganisms

The results of the most probable number (MPN) tests, and agar plate counts of leachate microorganisms from several landfills are shown in Table 3. The older landfills had lower counts and very low fecal-indicator bacterial levels. The counts of the Boone County Landfill, Cell 1(BC-1), were much lower than those found by Blannon and Peterson in 1974. The high eosin methylene blue levels in Table 3 were not always due to Gram-negative rods, but were due to *Bacillus* sp. or to fungal colonies. Moreover, the microbial levels indicated that the leachate (collected at the landfill surface) obtained from the fresh solid waste in the commercial landfill had a higher concentraton of Gram-negative rods and fecal streptococci in it than the older landfill leachate obtained from the pipes. These higher fecal-indicator bacterial concen-

TABLE 3. MICROBIAL COUNTS* FROM LANDFILL LEACHATE

Quantitative Tests	Landfills (Lysimeters)				
	Boone County		Center Hill	Commercial Landfill	
	BC-1 (6 years)#	BC-2 (5 Years)#	CH-19 (2 Years)#	From Pipes** (Years ?)#	From Landfill Surface*** (Weeks ?)#
MPN/100ml					
Total Coliforms	<30	<30	<30	<30	1.8×10^4
Fecal Coliforms	<30	<30	<30	<30	23
Fecal Streptococci	1×10^2	<20	80	50	3.5×10^4
Agar plate counts, CFU/100ml					
Blood-aerobic	7.3×10^5	5.0×10^4	1.7×10^5	5.0×10^5	6.2×10^6
Blood-anaerobic	8.0×10^5	$< 10^3$	1.8×10^4	1.9×10^5	1.3×10^5
Brain heart	4.0×10^5	1.0×10^4	6.0×10^5	1.8×10^4	2.7×10^6
Eosin Methylene Blue	3.0×10^5	1.0×10^4	9.0×10^3	6.0×10^3	5.0×10^6
Inhibitory Mold##	1.7×10^4	7.0×10^3	ND###	$< 10^3$	$< 10^3$
KF Streptococcal	$< 10^3$	$< 10^3$	7.0×10^2	5.0×10^2	2.5×10^3
Mycosel##	2.0×10^3	10^3	ND	$< 10^3$	$< 10^3$
Sabouraud##	4.0×10^5	2.0×10^6	2.8×10^4	3.0×10^5	$< 10^3$
Tellurite	5.0×10^4	6.0×10^3	7.0×10^3	5.0×10^2	$< 10^3$

*=These are the averages of two or more leachate samples.
**=The leachate came from pipes deep within the landfill ("old" leachate).
***=The leachate emerged from the soil at many sites ("fresh" leachate).
#=Age of waste in landfills and lysimeters.
##=Incubated at ambient temperature.
###=Not determined.

trations, indicated that fecal pollution was present in the landfill. Leachate from the commercial landfill pipes which extended deep within the old part of the landfill demonstrated low bacterial levels, whereas leachate arising out of the ground from recently deposited wastes contained microbial levels a thousand-fold higher.

The Inhibition of Bacteria by Leachate

The inhibition of the streptococci by the leachate in the lower dilutions of the fecal streptococcal MPN tests, described earlier, indicated that some toxicity was present. It seemed probable that the leachate toxicity could be responsible for decreasing plate counts and inhibiting some pathogen growth. To determine the presence and the extent of leachate toxicity, known bacterial species were exposed to leachate, and immediately plated out to determine the number of survivors. Selective agars were used to differentiate the test organisms from the leachate bacteria. The test bacteria used in this study were pathogens or pathogen indicator bacteria. They were *Klebsiella pneumoniae*, *Salmonella typhimurium*, *Pseudomonas aeruginosa*, and *Streptococcus faecalis*. These bacteria after exposure to the leachate had concentrations of 10^5/100 ml, while the controls, which were not exposed to the leachate, had levels of 10^7/100 ml or higher. These results demonstrated that the leachate inhibited the microbial growth of all four test bacteria. Furthermore, there is evidence in the literature to indicate that surpressed or inhibitory growth may also be a result of secondary stress caused by the selective media, in addition to the primary stress of leachate exposure [Postgate (20)].

Microbial Identification

The examination in the fall and winter months of leachates from the commercial and experimental field landfills revealed that the most frequently isolated microbes were *Bacillus* sp., *Streptococcus* sp., *Corynebacterium* sp., and yeast. In the early summer following the spring rains, numerous bacteria were found in all of the field landfill leachates, and these results are presented in Table 4. Some of the microorganisms in this table are known pathogens and can be found in fecal material. Many are saprophytes. All of these bacteria and fungi were compared to the U.S. Public Health Service's *Classification of Etiologic Agents on the Basis of Hazard* (21), which includes all disease-producing microorganisms. This classification is based on the pathology and infectivity of the microorganisms. Persons working with class one agents need take no special precautions in handling these microbes, whereas those working with class two agents need to take precautions to prevent from being inoculated by contaminated material containing these pathogens. Class three agents require that the investigator use special precautions to avoid direct contact with these pathogens, as contact will usually cause disease. Of the microrganisms found in Table 4, *Listeria monocytogens*, *Salmonella* sp., *Clostridium perfringens*, *Mycobacterium* sp., *Moraxella* sp., and *Acinetobacter* sp. are class two bacteria. No class three microorganisms were found in these studies. *Allescheria boydii*, while not on the etiologic list, is a pathogen which causes madura foot abscesses.

The *Mycobacterium* listed above could not always be found in leachate. so a concentration process was developed to improve the infrequent isolations. For this process one-half to one liter of leachate was filtered through diatomaceous earth. The diatomaceous earth was extracted with phosphate buffer, digested, centrifuged and inoculated onto Lowenstein-Jensen medium and spread on Middlebrook 7H10 agar plates. The *Mycobacterium* grew out on this medium within 2 to 3 weeks at 37 C. All eleven leachate samples produced *Mycobacterium* colonies. The levels isolated varied from 2 to 33/100 ml of leachate with an average of 10/100 ml. To separate these bacterial strains into groups for identification purposes, they were inoculated onto antibiotic media containing refampin, isoniazid, p-aminosalicylic acid, streptomycin and ethambutol. From the nineteen strains tested in this fashion seven sensitivity patterns emerged indicating possibly that seven different strains of mycobacteria had been isolated. Preliminary work has been started on their identification. In comparison with a known ATCC strain of *Mycobacterium fortuitum*, which grew out on Middlebrook 7H10 agar in 2 days, these isolated strains grew slower, taking just under a week to grow out. In addition, a yellow

TABLE 4. PARTIAL IDENTIFICATION OF MICROORGANISMS ISOLATED FROM SPECIFIC LEACHATE SAMPLES.

Leachate Source (Sampling dates)	Microorganisms Identified	
	Bacteria	Fungi
Boone County Cell 1. (5/3/78)	*Bacillus* sp. *Listeria monocytogens** *Aeromonas* sp. *Streptococcus* sp. *Moraxella* sp.* *Neisseria* sp. *Corynebacterium* sp.	Yeast *Allescheria boydii*
Center Hill Cell 19. (5/31/78)	*Pseudomonas* sp. *Streptococcus* sp. *Acinetobacter* sp. *Bacillus* sp.	Yeast *Fusarium* *Penicillium*
Commercial Landfill (6/12/79)	*Mycobacterium* sp.*	
Commercial Landfill (6/9/80)	*Proteus rettgeri* *Mycobacterium* sp.* *Salmonella* sp.* *Clostridium perfringens** *Citrobacter* sp. *Enterobacter aerogenes*.	

*=Listed as class 2 in the CDC *Classification of Etiologic Agents on the Basis of Hazard* (21)

strain was isolated a year ago, but the recent strains were cream colored. Mycobacteria are considered class two agents on the basis of hazard as described by the CDC classification, while *M. tuberculosis*, related to those isolated from the leachate, is considered a class three agent. This particular species has not been isolated from leachate, but there is every likehood that it is present because of the presence of other mycobacteria and high fecal-indicator levels in fresh landfill leachate samples.

RESULTS OF PHASE II EXPERIMENTS

Experimental Approach

This phase of the research investigated the survival of pathogenic microorganisms in leachates from experimental landfills (lysimeters) containing municipal solid waste, sewage sludge, hospital waste, and combinations of municipal solid waste and sewage sludge or hospital waste. Sewage sludge and hospital waste are produced in great quantities in our communities, are potentially hazardous, and are frequently placed into landfills throughout the United States. The impact of all of these wastes on the disease potential of landfill leachates is not known. Landfills containing such wastes could produce leachates carrying pathogens into the environment via the groundwater or surface waters. It is, therefore, critical to know the duration of pathogenic microbial survival in landfills and their leachates. Accordingly, the lysimeter studies of Phase II were designed to evaluate the survival characteristics of pathogenic microorganisms and their indicators. Of special interest was the speciation of streptococcal isolates, because they represented a different type of pathogen-indicator in that the source of the waste, i.e. from animal, human, avian, plant, or insect sources, could also be ascertained.

Finally, the examination of buried solid waste from old landfills or lysimeters for the presence of pathogens and pathogen-(fecal)-indicators would show the extent of their survival and their potential for groundwater and surface water pollution.

The Enumeration and Identification of Microbes Found in Solid Wastes

Samples of waste placed in the six lysimeters were first analyzed for their microbial numbers. Identification of the microbes isolated included streptococcal speciation. Densities were determined by agar plate counts and the most probable number (MPN) procedures. The results, using several agar plate counts including aerobic and anaerobic blood, EMB, and Sabouraud were similar, i.e. about 10^8 microorganisms per gram of waste. Sewage sludge samples were characterized using KF-Streptococcal and tellurite agars. Plate counts on both media were 10^5/gram of sludge. The hospital waste microbial levels using these last two media were 1 to 2 logs higher than those found with sewage sludge, whereas the municipal waste was 3 logs higher. All three of these wastes contained total coliforms, fecal coliforms, and fecal streptococci at levels 10^7 to 10^9 MPN/g of waste. The pathogens present in these wastes were identified as *Klebsiella* sp., *Salmonella* sp., *Mina* sp., *Herellea* sp., *Moraxella* sp., and *Pasteurella hemolyticum*. Of the 25 pathogens isolated from these wastes, 6 came from sewage sludge, 9 from the hospital waste, and 10 from the municipal waste.

Microbial Densities in Leachates

The determination of plate counts and MPN levels of microorganisms in the six lysimeter leachates were made weekly, biweekly, monthly and then four times a year. Although the data extends from the first leachate sample, which appeared in some cases at the 4th week after lysimeter construction, to the 104th week (two years), only the concentration levels for the 13th and the 104th weeks, the latter placed in parenthesis, are shown in Table 5. The Gram-negative rod concentrations, which had been high in the solid waste, had dropped to a low level by the 13th week. Furthermore the results taken at the 104th week show even lower microbial levels. The fecal-indicators were gone (below the detection level); most or all of the aerobic and anaerobic blood agar, Sabouraud and tellurite plate counts were lower; while the Standard Methods agar levels were unchanged. Thus, the Gram-negative rods (e.g. *Salmonella*, *Klebsiella*, other coliforms, etc.) died within the first three months, while the streptococci were < 10/100 ml at the end of the first year. In addition, the nonsporing Gram-positive rods and fungi had decreased in concentration.

Other assays not shown here demonstrate that Gram-positive sporeforming rods such as *Clostridium perfringens* and *Bacillus* sp. survived. This population change over the 2 years indicates that many pathogens and pathogen-indicators present in the solid waste had died in the lysimeters. However, not depicted by these results was the occasional release of leachate from the sewage sludge lysimeter containing relatively high levels of fecal-indicator bacteria, such as 2.8 x 10^3, 4.9 x 10^1, and 5.4 x 10^3 MPN/100 ml of total coliforms, fecal coliforms, and fecal streptococci, respectively, after 2 ¼ years.

The low concentration of fecal-indicator bacteria in leachates from most lysimeters after 3 months and certainly after 2 years suggest that these bacteria die quickly in landfills. However, the survival and continued discharge of high concentrations of fecal-indicator bacteria in leachates from the sewage sludge lysimeter 2 years after its construction demonstrates the potential for intermittant contamination by this waste for an extensive time period.

Total Coliform & Fecal Coliform Densities

The total coliform and fecal coliform MPN levels in the leachate from all lysimeters had disappeared by the 9th week after lysimeter construction, indicating that all the coliforms had died. However, the substitution at the 13th week of the *Standard Methods* (1) MPN test to the Allen et al. (2) modification indicated that fecal and/or total coliform MPN levels were indeed present in leachates from both lysimeters A and B. These lysimeters contained sewage sludge, and sewage sludge plus municipal waste, respectively. The total coliform MPN levels for lysimeter A reached a peak of 2.2 x 10^2/100 ml at the

TABLE 5. MICROBIAL CONCENTRATIONS IN LEACHATES OBTAINED FROM THE LABORATORY LYSIMETERS AT THE 13th AND THE 104th WEEK AFTER CONSTRUCTION

Quantitative Tests	UC Laboratory Lysimeters*					
	A	B	C	D	E	F
MPN/100 ml						
Total coliform	20 (<2)#	86 (<2)	40 (<2)	<20 (<2)	<20 (<2)	<20 (<2)
Fecal Coliform	<20 (<2)	<20 (<2)	<20 (<2)	<20 (<2)	<20 (<2)	<20 (<2)
Fecal Streptococci	4.9×10^2 (<2)	1.6×10^4 (<2)	3.5×10^2 (<2)	2.3×10^2 (<2)	2.9×10^4 (<2)	7.9×10^2 (<2)
Agar Plates, CFU/100 ml						
Blood-aerobic	1.9×10^7 (1.2×10^8)	1.2×10^8 (1.3×10^6)	6.8×10^7 (1.6×10^6)	2.9×10^8 (1.4×10^7)	7.6×10^6 (9.5×10^5)	1.2×10^7 (3.3×10^5)
Blood-anaerobic	6.3×10^5 (1.2×10^8)	1.1×10^7 (3.3×10^6)	8.0×10^7 (3.2×10^6)	5.6×10^7 (4.0×10^7)	3.1×10^7 (1.7×10^6)	2.1×10^8 (2.7×10^6)
Standard Methods	1.4×10^8 (1.0×10^8)	5.4×10^7 (1.4×10^6)	7.9×10^7 (1.2×10^8)	3.1×10^8 (2.2×10^7)	1.5×10^7 (2.8×10^7)	6.3×10^8 (3.1×10^4)
Sabouraud	2.2×10^7 (3.1×10^6)	4.9×10^7 (4.2×10^5)	2.2×10^7 (4.2×10^5)	6.6×10^7 (1.7×10^6)	4.6×10^7 (3.5×10^5)	5.2×10^6 (1.9×10^4)
Tellurite	1.1×10^6 (4.0×10^6)	1.6×10^6 (2.9×10^4)	7.8×10^6 (4.0×10^3)	4.5×10^7 (1.6×10^4)	1.7×10^6 (6.4×10^4)	5.9×10^6 ($< 10^3$)

*=The content of lysimeters are: A, sewage sludge; B, municipal solid waste plus sewage sludge; C, municipal solid waste; D, municipal solid waste; E, municipal solid waste plus hospital waste; and F, hospital waste.

#=The microbial counts after 104 weeks are in the parentheses.

15th week and had measurable levels lasting through the 35th week. The lysimeter B leachate contained total coliform levels of 8.0 x 10^2/100ml at the 13th week, and decreased to < 2/100 ml thereafter. MPN levels in leachates from lysimeters C and F remained at < 20/100 ml and did not change. Thus, the two lysimeters containing sewage sludge continued to show coliforms in their leachates for longer periods of time than those lysimeters without sewage sludge. As described above, lysimeter A continued to yield fecal bacteria for 2 years after its construction. The very sensitive procedure of Allen et al.(2) detected stressed or small numbers of bacteria, while the less sensitive procedure for Standard Methods (1) did not allow their growth or their detection.

The Identification of Streptococci Groups and Species

Blannon and Peterson (3) suggested that the speciation of streptococci isolated from landfill leachate would indicate the natural origin of the streptococci, and this in turn would indicate the origin of the solid waste. The natural origins of all streptococcal species are presented in Table 6.

The identified streptococci from the lysimeters are shown in Table 7. The first group of streptococci identified here were isolated from the wastes added to the lysimeters when they were constructed, and from the six lysimeter leachates. The lysimeter wastes, both municipal and hospital, contained equal numbers of Group D and viridans streptococci. Most of the Group D were identified as Streptococcus faecalis subsp. liquefaciens, which is of plant or insect origin. This latter microbe has no health significance. On the other hand, the viridans group does come from man, and this indicates the human source of the waste.

Streptococci isolated from the lysimeter leachates were similar to those found in the solid waste. Most of the streptococci from Lysimeter A were viridans, which originates from man. Lysimeter B leachates contained S. faecium and S. durans. These both originate from the feces of man and animals. Some of the streptococci isolated from the leachates produced by municipal solid waste lysimeters C and D were Group D streptococci, but not enterococci. These cocci probably came from animal feces , such as pets, horses and cows.

A year later additional grouping and speciation were done on the surviving streptococci from these lysimeters and landfills. The results are shown in Table 8. After a year and a half, the B and E lysimeters still retained viable human or animal Streptococcus faecalis, as did the Boone County lysimeters landfill after 8 years. The fecal origin of the commercial landfill is also evident from the presence of large numbers of Group D enterococci, Streptococcus faecalis, and viridans streptococci. Since these streptococci were able to survive for 8 years under landfill conditions, so may pathogens of equal resistance.

The final experiment on the six laboratory lysimeters was the determination of surviving pathogen and fecal-indicator concentrations in the solid waste within each lysimeter. Two lysimeters have been examined, although the results from the D lysimeter only is presented in Table 9. The waste samples were taken at 10 cm intervals. It may be recalled that the leachate from D lysimeter after 2 years (Table 5) had no fecal coliforms and only 10^2/100 ml fecal streptococci. Yet in the solid waste within the lysimeter, measurable levels of the fecal-indicators were obtained. The survival of these bacteria were related to the pH of the solid waste, as the microbial counts decreased with an increase in acidity. In addition, the Boone County experimental landfill's solid waste was analyzed after 9 years at various levels. Four soil controls from a nearby site were taken at the same depth levels as those from the landfill, and analyzed in a similar manner. The results of these studies in Table 9 demonstrate that the solid waste's coliform and streptococci levels in the landfill differ significantly from those in the control sod samples. Moreover, the solid waste and leachate in the bottom part of the landfill had significant levels of fecal-indicator bacteria present. Apparently, these bacterial strains had survived in the waste for 9 years. It will be recalled from Table 3 that the fecal-indicator levels in the leachate after 6

TABLE 6. THE HUMAN, ANIMAL & VEGETABLE ORIGINS of STREPTOCOCCAL GROUPS

Group*	Species	Origin#
A	*S. pyogenes*	man
B	*S*. agalactiae	cattle, man
Not Groups A, B or D (i.e., Groups C, G, F)	*S*. dysgalactiae	cattle
	S. equi	horse
	S. equisimilis	man, animal
	S. zooepidemicus	animal
	S. anginosus	man
D, not enterococcus	*S. bovis*	animal
	S. equinus	horse
Enterococcus (Group D)	*S. faecalis*	man, animal
	S. faecalis subsp. *liquefaciens*	plants insects
	S. faecalis subsp. *zymogenes*	man, animal
	S. faecium	man, animal
	S. durans	man, animal
Viridans	*S. salivarius*	man
	S. mitis	man
Q	*S. avium*	fowl

*Lancefield Groups are A through Q.

#From Wilson & Miles (26), except for *S. faecalis* subsp. *liquefaciens* which is from Geldreich, et al. (10).

TABLE 7. STREPTOCOCCAL GROUPS[*] AND SPECIES[*] ISOLATED FROM SOLID WASTE USED IN THE LYSIMETERS AND FROM THE LEACHATE AFTER CONSTRUCTION

Streptococcal Groups	Number of Streptococcal Isolates From:							
	Municipal Solid Waste	Hospital Solid Waste	Lysimeters[**]					
			A	B	C	D	E	F
Group A	#	#	#	#	#	#	#	#
Group B	1	#	#	#	#	#	#	#
Not Groups A, B or D (i.e., Group C, G and/or F)	2	1	#	#	#	#	#	#
Group D (enterococcus)								
S. faecalis subsp. *liquefaciens*	3	5	#	#	#	#	9	10
S. faecalis	1	#	1	1	#	#	3	8
S. faecium	1	#	4	9	4	#	7	#
S. durans	#	#	#	7	3	#	1	4
TOTAL Group D	5	5	5	17	7	#	17	22
Group D (not enterococcus)	1	2	2	#	5	6	4	1
Viridans Group	3	4	6	2	#	#	1	1
S. salivarius	2	1	#	#	#	#	#	#
TOTAL Viridans Group	5	5	6	2	#	#	1	1

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[*] The streptococcal groups were identified by the procedure of Facklam, et al. (9) and the species were identified by the procedure of Pavlova, et al. (18).

[**b] The content of lysimeters are: A=sewage sludge; B=municipal solid waste plus sewage sludge; C=municipal solid waste; D=municipal solid waste; E=municipal solid waste plus hospital waste; F=hospital waste.

[#] None detected.

TABLE 8. THE PRESUMPTIVE IDENTIFICATION OF STREPTOCOCCAL GROUPS* AND SPECIES** ISOLATED FROM LANDFILL LEACHATES.

Streptococcal Sources	Sensitivity to Bacitracin	Hippurate Test	Growth on: Bile Esculin	Growth on: 6.5% NaCl	Isolates: Number	Isolates: Identification
Lysimeter Leachates:						
UC-E	-	-	+	+	14	D-enterococcus (6 *S. faecalis*)
UC-B	-	-	+	+	2	D-enterococcus (2 *S. faecalis*)
BC-2	-	-	+	+	12	D-enterococcus (12 *S. faecalis*)
CH-14	-	-	+	+	10	D-enterococcus
Commercial Landfill Leachates (CL)#:						
CL, Sample 1	-	-	+	+	16	D-enterococcus (15 *S. faecalis*)
CL, Sample 2	+	-	-	+	2	D-enterococcus (2 *S. faecalis*)
	-	-	-	+	2	D-enterococcus
	+	-	-	-	3	Viridans
	-	-	-	-	11	D-enterococcus
CL, Sample 3	-	-	-	+	7	D-enterococcus
CL, Sample 4	-	-	-	-	3	Viridans
CL, Sample 5	-	-	+	-	2	D-nonenterococcus
	-	-	-	+	2	D-enterococcus
	-	-	+	+	4	D-enterococcus
Streptococcus faecalis ATCC 19433	-	-	+	+	1	D-enterococcus
			Total		91	

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*=Facklam *et al*. (1974)

**=Pavlova *et al*. (1962), *Stanard Methods* (1976), Bordner *et al*. (1978)

#=Obtained from the surface of the landfill slope.

TABLE 9 FECAL INDICATOR BACTERIAL PRESENCE IN MUNICIPAL SOLID WASTE FROM A TWO YEAR OLD LABORATORY LYSIMETER AND A NINE YEAR OLD LANDFILL

Type of Sample	Samples Collected At These Descending Levels	MPN/100 grams of Solid Waste* and soil or MPN/100 ml Leachate			pH of Sample
		Total Coliform	Fecal Coliform	Fecal Strepto-coccus	
UC Lysimeter D					
Solid Waste	10 cm	1.3×10^3	9.0×10^1	7.1×10^4	7.1
Solid Waste	20 cm	2.2×10^3	< 20	2.4×10^3	6.2
Solid Waste	30 cm	2.0×10^1	< 20	2.0×10^1	5.2
Leachate	40 cm	< 2	< 2	< 2	5.2
Boone County Landfill BC-1					
Top Soil	46 cm (1½ ft.)	2.4×10^5	3.5×10^3	9.8×10^5	7.0
Solid Waste	152 cm (5 ft.)	1.1×10^2	< 20	2.0×10^4	6.0
Solid Waste	214 cm (7 ft.)	2.4×10^3	2.0×10^1	6.0×10^4	5.4
Solid Waste	259 cm (8½ ft.)	1.6×10^5	2.3×10^2	3.3×10^5	7.4
Solid Waste	274 cm (9 ft.)	5.4×10^4	3.3×10^2	ND**	7.6
Leachate	304 cm (10 ft.)	9.2×10^3	3.5×10^2	1.1×10^2	7.0
Decomposed Waste at Plastic Liner	320 cm (10½ ft.)	3.5×10^4	4.9×10^2	3.5×10^4	6.9
Control-Soil near Boone County Landfill					
Top Soil	46 cm (1½ ft.)	$\geq 2.4 \times 10^5$	< 20	2.1×10^3	5.7
Soil	152 cm (5 ft.)	2.3×10^2	< 20	2.3×10^2	5.4
Soil	213 cm (7 ft.)	5.0×10^1	< 20	< 20	7.0
Soil	259 cm (8½ ft.)	5.0×10^1	< 20	5.0×10^1	7.7

*=Wet weight
**=Not determined

years had no total and fecal coliform bacteria present but had 10^2/100 ml of fecal streptococci. However, the high levels of microbes in the Boone County landfill solid waste suggests that fecal-indicator bacteria, and perhaps pathogens related to them, survive for longer periods of time. Finally, the fecal-indicators survived in higher concentrations in the landfill than in the lysimeters. This may indicate a difference in the waste storage conditions, such as the volume of waste, temperature, humidity, pH, age of the waste, etc.

CONCLUSIONS

This work shows that some pathogenic bacteria have been isolated from landfill leachate, and suggests that with improved methods, more may be found. In addition, leacnate from a 2 year old sewage sludge lysimeter continued to produce leachate intermittently, containing significant levels of fecal-indicator bacteria. Finally, these same bacteria were demonstrated in municipal solid waste and hospital waste from lysimeters that were 2 years old, and also from a fullscale landfill that was 9 years old. These results demonstrate that fecal-indicator bacteria do not die as rapidly as previously reported, but survive for extended periods of time with the potential to contaminate ground and surface waters when released from the landfill.

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WASTEWATER TREATMENT PLANT RESIDUAL LANDFILLING: A CRITICAL REVIEW

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ABSTRACT

A literature review was conducted to critically evaluate the landfilling of wastewater Treatment Plant (WTP) residuals. Current practices, adverse environmental and public health impacts, beneficial uses, available control technology and management options are described and evaluated. Existing literature revealed a lack of information regarding the environmental and public health impacts related to WTP residual landfilling. Epidemiological information regarding the effects of WTP residual landfilling was absent. WTP residual/refuse co-disposal and sludge-only narrow trench alternatives represent the most environmentally acceptable of the landfilling methods. For the most part, co-disposal practices pose greater environmental public health risks than conventional municipal refuse landfills, the extent of which is largely a factor of residual treatment and landfill management. Apart from organic and microbiological components, leachates from WTP residual landfills (alone or co-disposal) are not unlike municipal refuse landfill leachates. It is apparent that the practice of residual disposal in municipal refuse landfills is not only cost-effective, but can be properly managed to provide protection of the environment and public health.

INTRODUCTION

Disposal of Wastewater Treatment Plant (WTP) residuals have historically occurred by:

1. Various land burial procedures
2. Direct discharge to oceans
3. Internment in evaporative (or percolation) type lagoons and ponds
4. Incineration
5. Agricultural utilization
6. Spray irrigation
7. Land and/or forest application

Bans on ocean dumping of residuals and the high costs of incineration have stimulated interest in landfilling. Recent amendments to the Clean Water Act and Solid Waste Act require that landfilling of these residuals, either alone or with refuse, does not pose an unreasonable risk of adverse environmental or public health impacts.

The purpose of this study was to critically evaluate the landfilling of WTP residuals for the purposes of:

- a) describing current disposal practices
- b) determining adverse/environmental and public health impacts
- c) describing available control technology
- d) evaluating management options

Landspreading and beneficial use of sludge, such as for agricultural uses, was not considered in this evaluation.

Information from a variety of sources was selected on the basis of its significance and relevance. These sources included a computer search of 15 commercially available data bases, general references, solid waste management representatives, and landfill sites personnel where studies had been conducted. Where information was deficient, conclusions were drawn based on experience, scientific judgement, and inference, and are subject to testing and confirmation.

RESULTS AND DISCUSSION

Types of Residuals

Since the landfilling of WTP residuals is primarily constrained by their physical properties (e.g. moisture content) and environmental/public health considerations, this report will classify the residuals according to those treatment processes which represent distinct changes in the physical, chemical, or pathogenic nature of the residual.

In general, residuals can be characterized according to whether they are treated or untreated. Untreated residuals are represented by:

- grit, screenings, and floatables
- raw, primary sedimentation sludge
- raw, waste activated sludge

The extent to which a residual is stabilized prior to landfilling may have a pronounced affect upon leachate and gas generation and composition, the pathogenic nature of the residual, the quantity and physical properties of the residual, and residual odor. Advantages and disadvantages of the major stabilization process include:

1. Anaerobic and aerobic digestion-reduces total sludge mass and thus reduce cost of disposal; reduces pathogens; produces a sludge of relatively poor mechanical dewatering characteristics.
2. Lime treatment-reduces odor and odor production potential in sludge; reduces pathogen levels; improves dewatering and settling characteristics; lowers concentrations of soluble phosphate, ammonia nitrogen and total Kjeldahl nitrogen, and heavy metals.
3. Chlorine oxidation-inactivates pathogens; reduces odor; improves dewaterability of digested sludge; chlorine-stabilized sludges pose substantial risks of highly toxic leachates because of its high acidity and potential for releasing chlorinated hydrocarbons and heavy metals.
4. Heat treatment-inactivates pathogens and can reduce sludge mass; heat treatment cannot reduce odor.

Because of potential leachate generation, landfilling of WTP residuals usually requires that residuals be dewatered prior to disposal. Dewatering residuals from a 5 to 20 percent solids concentration reduces volume by three-fourths and results in a damp solid. Various residual conditioning methods may be employed which are designed to improve the dewaterability of WTP residuals, and can provide disinfection, affect odors, alter the wastewater solids destruction or addition. Common methods include:

1. Inorganic chemical conditioning--the addition of lime, ferric chloride, or ferrous sulfate as flocculant aids; inorganic chemical conditioning increases the sludge mass by 15 to 30 percent, but can be beneficial because of its stabilization effects.
2. Polyelectrolyte conditioning--little additional sludge mass is produced
3. Thermal conditioning--the solids concentration of the cake and thus its dewaterability is nearly double that of the previous two conditioning alternatives; process sterilizes the residual.
4. Elutriation--the process of washing the alkalinity out of anaerobically digested sludge, improving its dewaterability.

High temperature processes, such as incineration or wet air oxidation, significantly reduce residual volume and weight (95%), thereby, reducing disposal requirements. Additionally, these processes can destroy or reduce toxics that otherwise create adverse environmental impacts.

Current Practices of WTP Residual Landfilling

Landfilling approaches for WTP residuals are usually classified according to three major categories:

- Residual-only Trench

Narrow Trench
Wide Trench

- Residual-only Area Fill

  Area Fill Mound
  Layer Fill
  Diked Containment

- Co-disposal

  Residual/Refuse Mixture
  Residual/Soil Mixture

The residual-only trench method involves the excavation of a trench into which the residual is placed. The excavated soil is subsequently placed over the residual as a cover. Area fill methods involve residual placement upon the original ground surface. In general, area fills require residual mixing with a soil bulking agent in order to provide sludge stability. Residual-only area fills are constructed in locations where a high groundwater table or near-to-surface bedrock conditions prevail. Cover soil needs place additional demands upon onsite or imported soil needs.

Co-disposal is the practice of depositing residuals at a conventional sanitary landfill. Co-disposal practices offer significant cost savings due to the shared capital and operational expenses with sanitary landfills (1 , 2 ,25). Co-disposal of soil and residual is used as a cover for municipal refuse landfills.

POTENTIAL ENVIRONMENTAL IMPACTS

Impacts on Water Quality

Because of the toxic and sometimes pathogenic nature of WTP residuals, its placement into a landfill may lead to the movement of highly contaminated leachates out of the fill and into the surrounding soil and subsurface aquifers. Figure 1 depicts the concentration ranges of various chemical and biological constituents of leachates from (1) municipal refuse (2) admixed municipal refuse and WTP residuals and (3) WTP residuals alone. In the latter cases, the residuals were either untreated or stabilized. Segregation of disposal methods according to types of residuals was not attempted since many studies failed to indicate the degree of residual treatment, and since many landfills accepted assorted residuals. Documented investigations of WTP residual landfill leachate composition are minimal, considerably less than for municipal landfills. The major research was performed by Lofy et al. (11), Stone (24), Johansen et al. (9), Emcon and Associates (4), Pickard (16), Walker et al. (26), Charlie et al. (3), and Sikora et al. (20).

Figure 1 indicates that concentration ranges of leachates constituents of municipal refuse landfills and WTP residual/refuse co-disposal are similar for the major organic indicators, total solids, nutrients, major ions, and pathogens. Maximum concentrations of heavy metals and pH, however, differ. Leachates of municipal refuse exhibit a lower minimum pH, and higher maximum concentrations of Fe, Zn, and Cd than co-disposal methods. This evidence indicates a possible buffering effect provided by WTP residuals with regard to municipal refuse leachates, decreasing the solubility of various heavy metals.

With the exception of pH, COD and certain heavy metals, Figure 1 shows no significant difference in the quality of leachates of residual only and residual/refuse co-disposal landfills. The pH of the residual only disposal method was found to exist within a relatively narrow range (6.2 - 8.3) in comparison to residual/refuse co-disposal methods (4.5 - 10). Interestingly, maximum concentrations of iron and zinc in residual only leachates are nearly one order of magnitude less than co-disposal leachates. It should be noted that comparatively few studies are represented in the WTP residual leachate data; further data is necessary to verify these results.

Concentrations of WTP residual leachate constituents, whether landfilled alone or with municipal refuse, pose adverse impacts to the beneficial uses of receiving waters. Of primary concern are specific organics (e.g. toxic refractories such as PCB, pesticides, etc.), heavy metals, and pathogens. Table 1 identifies the potential groundwater contaminants associated with various common WTP residuals, assuming improper landfill management. A comprehensive review of the water quality impacts posed by WTP residual leachates is presented by Lu et al. (12).

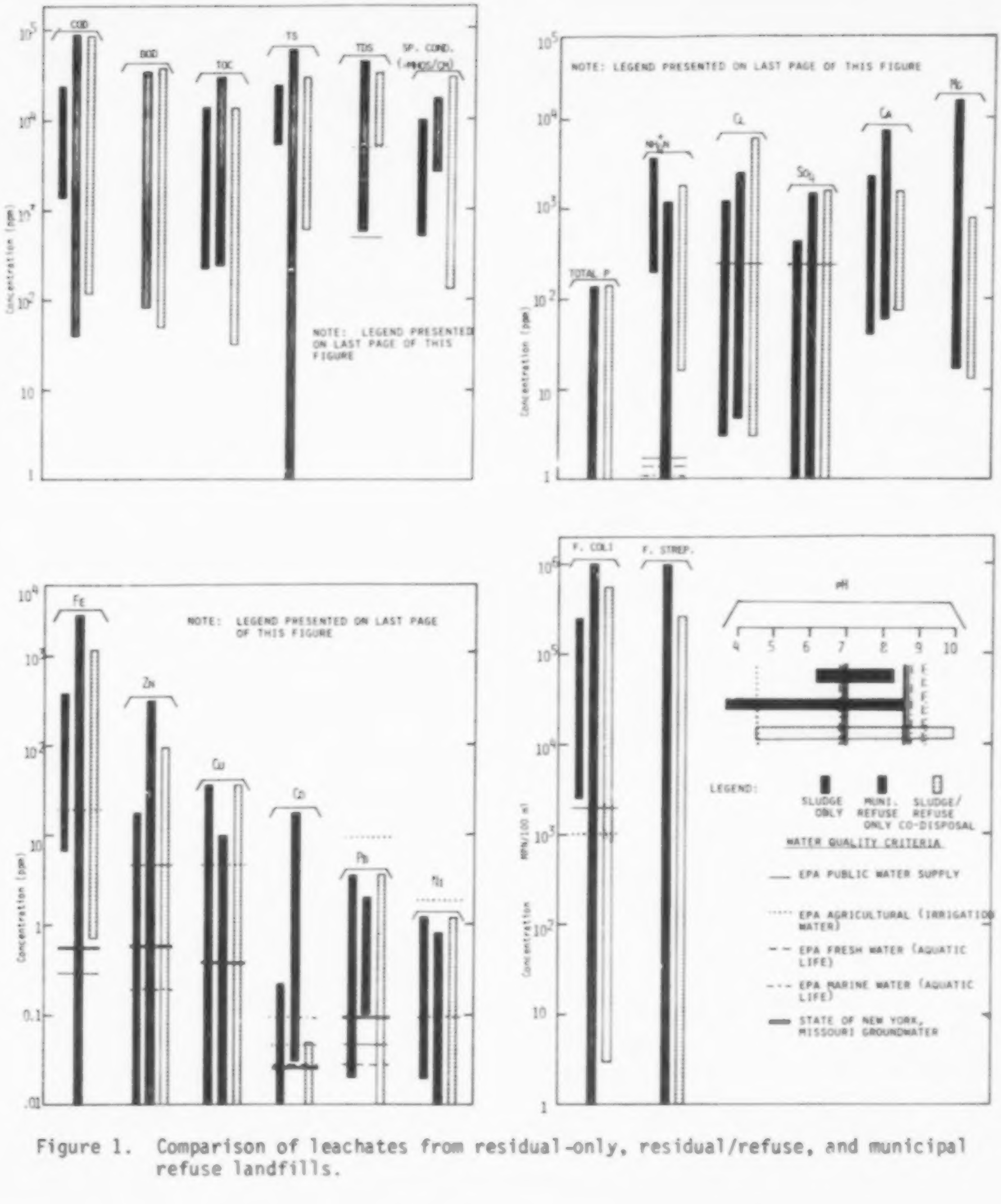


Figure 1. Comparison of leachates from residual-only, residual/refuse, and municipal refuse landfills.

TABLE 1. POTENTIAL GROUNDWATER CONTAMINANTS OF VARIOUS WTP RESIDUALS*

Type of Residual	Contaminants
Liquid - unstabilized	
Gravity thickened primary, WAS and primary, and WAS	Org.,HM, TDS, NO_3, Path.
Flotation thickened primary and WAS, and WAS without chemicals	Org.,HM, TDS, NO_3, Path.
Flotation thickened WAS with chemicals	Org., HM, TDS, Path.
Thermal conditioned primary or WAS	Org., HM, TDS, NO_3
Liquid - stabilized	
Thickened anaerobic digested primary and primary, and WAS	HM, TDS, NO_3
Thickened aerobic digested primary and primary, and WAS	HM, TDS, NO_3
Thickened lime stabilized primary and primary, and WAS	HM, TDS
Dewatered - unstabilized	
Vacuum filtered, lime conditioned primary	Org., HM, TDS, NO_3, Path.
Dewatered - stabilized	
Drying bed digested and lime stabilized	TDS
Vacuum filtered, lime conditioned digested	HM, TDS, NO_3
Pressure filtered, lime conditioned digested	HM, TDS, NO_3
Centrifuged, digested and lime conditioned digested	HM, TDS, NO_3
Heat dried	
Heat dried digested	HM, TDS, NO_3
High temperature processed	
Incinerated dewatered primary and primary, and WAS	HM
Wet-air oxidized primary and primary, and WAS	Org., HM, TDS, NO_3, Path.

* Resulting from improper WTP residual landfill management
WAS - Waste-activated sludge
Org.- Organics
HM - Heavy Metals
TDS - Total Dissolved Solids
NO_3 - Nitrate-nitrogen
Path. - Pathogens

Impacts on Air Quality

Noxious gases, including H_2S and various mercaptans, are commonly associated with WTP residuals and represent a primary source of adversity with regard to residual disposal near population centers. Like sanitary landfills, carbon dioxide and methane comprise as much as 90 percent of the gases produced at residual/refuse co-disposal sites.

Volatilization of organic constituents can occur at residual landfill sites. The vaporized contaminants of most concern, due to their toxicity, are halogenated organics and aromatic hydrocarbons. Aside from pesticides, available information on the volatilization of these compounds is limited. Spencer and Claith (23) have suggested that volatilization of surface deposits and soil incorporated pesticides plays a major role in the dissipation of pesticides from solid surfaces. The potential volatibility of pesticides and other organic compounds is related to the inherent vapor pressure of the compound, but the actual volatilization will depend upon environmental conditions. Shen and Tofflemire suggest that at least 99% reduction in volatilization can be permamently achieved by a low permeability topsoil cover (19).

Effects of aerosols from WTP residual landfills are unknown. However, it has been determined that airborne bacteria and virus-containing particles may be generated from sewage treatment plants and spray irrigation operations (7,10,15,21,22). Therefore, the application or discharge of WTP residuals at landfill sites may be a source of viable pathogenic aerosols.

Impacts on Land Quality

Landfilling of WTP residuals may be constrained by a number of land or soil related impacts. Among these include:

- Certain disposal areas may be particularly sensitive to groundwater contamination from leachates, or may provide a habitat for rare wildlife .
- Clearing of land necessary for landfill construction or daily operation may accelerate erosion.
- Surrounding soils may accumulate high concentrations of contaminants making the soil unfit for future beneficial uses.

An extensive review of areas critical for WTP residual disposal to land due to environmental hazards or other limitations is presented by Schmidt et al. (17).

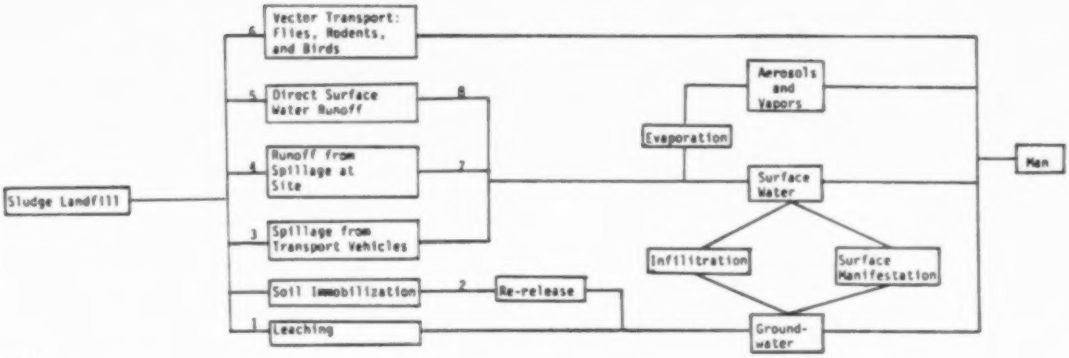
POTENTIAL PUBLIC HEALTH IMPACTS

Potential public health impacts connected with WTP residual landfilling operations can best be defined by: (1) identifying the residual contaminants which represent threats to public health, and (2) identifying the pathways through which the contaminants associated with WTP residual landfilling can intersect with man. Residual contaminants posing direct threats to public health can be classified as follows:

- General water quality contaminants (particularly nitrate)
- Elemental contaminants
- Biocidal contaminants
- Synthetic/organic contaminants (non-biocides)
- Biological contaminants

Traditionally biological contaminants have generated the most interest in WTP residual handling and disposal since these contaminants may be directly responsible for infecting human populations. Considerable research has been done with regard to the remaining contaminants and their potential for impairing human health, and is reviewed by Lu et al. (12).

The pathways available for disease transmission include atmospheric, surface water and groundwater routes. A simplified pathway chart is presented in Figure 2 . The leaching of contaminants from the landfill to underlying groundwater poses perhaps the greatest risk to public health, insofar as WTP residuals and leaching effects are far more difficult to manage once final placement has occurred. Contaminant transport through the atmosphere via aerosols and vapors is speculated, though the only direct evidence of such is in regard to biological contaminants


Vector Transport: Flies, Rodents, and Birds
Direct Surface Water Runoff
Runoff from Spillage at Site
Spillage from Transport Vehicles
Soil Immobilization
Leaching
Sludge Landfill
Re-release
Evaporation
Aerosols and Vapors
Surface Water
Infiltration
Surface Manifestation
Ground-water
Man
1
2
3
4
5
6
7
8

LEGEND

1. Liner (artifical or synthetic); impermeable cover soil surface water control
2. Maintenance of alkaline pH
3. Safety locks on trucks, closed vehicles
4. Surface water control measure: i.e., diversion ditches, etc: treatment with lime
5. Same as 4: installation of cover soil; revegetation
6. Cover soil disinfectant stockpiled sludge reduction of exposure of sludge
7. Collection and treatment/recycle; utilize overland flow principles for treatment
8. Same as 7.

Figure 2. Simplified contaminant pathways chart and commonly employed control measures.

generated by municipal wastewater processing systems (7,10,15,21,22).

WTP residual landfill design and operational procedures govern the availability of pathways for contaminant transmission. Trenching of residuals provides several inherent advantages toward minimizing contaminant transport from disposal sites. These include:

- Absence of premixing with soil reduces available contact time with the environment
- Stockpiling is seldom employed
- Erosion of residual is less pronounced due to its placement below ground level.

Conversely, area fill operations (area fill mound, area fill layer, and diked containment) have several design features conducive to contaminant transmission via surface runoff. These include:

- Continuous slumping and mound movement
- Erosion of mound or dike failure (in the case of diked containment)
- Potential of runoff from stockpiled residuals during wet weather periods.

WTP residual/refuse co-disposal operations pose risks of contaminant transmission via surface runoff due to the logistics of refuse and residual arrival at the site (i.e. stockpiled residuals).

Available Control Technology

Measures to control the transport of pollutants from WTP residual landfills are presented in Figure 2. Many of the control measures are presented in detail in the Process Design Manual for Municipal Sludge Landfills (18). Leachate control, including leachate volume control, collection, and treatment is the subject of much recent research, particularly for sanitary landfills. Because WTP residual only landfill and co-disposal landfill leachates are of similar composition to municipal landfill leachates, control and treatment measures should apply in either case. A comprehensive review of leachate control and treatment measures is provided by Lu *et al.* (13).

Management Options

The most significant features governing the selection of a WTP residual landfilling method are:

- Residual percent solids
- Residual characteristics (stabilized or unstabilized)
- Hydrogeology (deep or shallow groundwater or bedrock)
- Groundslope

Because landfilling of WTP residuals is constrained by environmental impacts, public health impacts, costs, and the availability of land, the selection process must estimate the potential magnitude of each of these factors with regard to each landfilling alternative. Table 2 shows the results of these efforts. Of the WTP residual landfilling alternatives, the residual only trench and residual/refuse co-disposal methods represent significant advantages in terms of costs, environmental factors (odors, vectors, and aerosols), potential public health impacts, and potential for future land use at disposal sites.

Cost rankings in Table 2 were based on total site costs, i.e. capital plus operating costs. Co-disposal methods can, in some instances, provide substantial cost savings in residual disposal inasmuch as dewatering may not be required. Stone (24) estimated that 0.6 to 1.8 lbs of liquid can be added for every 1.0 lb of dry weight solid waste before complete saturation is reached. Based on an initial refuse moisture content of 65 percent (which represented a "worst case" condition in Stone's study), the maximum possible mix ratios for residual/refuse co-disposal methods are listed in Table 3 . These values indicate the quantity of residual that can be applied to municipal refuse without exceeding its moisture retention capacity. Of course, rainfall and other sources of moisture will reduce the absorbtive capacity of refuse and may significantly decrease the ratios indicated. Field testing is necessary to verify these values.

Residual dewatering is imperative to many landfill designs and operations from a construction standpoint. Table 4 outlines the important physical characterstics of residuals of different solids content in terms of workability. Other residual characteristics which determine suitability for landfilling include:

- The anticipated composition and strenth of leachates
- The anticipated rate and volume of leachate production
- The anticipated composition and volume of gas produced
- The extent of stabilization
- Residual settlement

TABLE 2. CRITERIA INFLUENCING THE SELECTION OF A RESIDUAL LANDFILLING PRACTICE

Current WTP Residual Landfilling Practices		Land Required	Soil Bulking Agent Required	Residual Solids Content Recommended (%)	Costs	Environmental Factors					Leachate Control Measures Required	Potential Public Health Impacts	Potential For Future Land Use At Disposal Site
						Odors	Vectors	Noise	Fire Hazard	Aerosols			
Residual Only	• Residual Only Trench												
	Narrow Trench	H	No	≥ 15	M	L	L	M	L	L	L	L	M
	Wide Trench	M-L	No	≥ 20	M	L	L	H	L	M	M	M	L
	• Residual Only Area Fill												
	Area Fill Mound	M-L	Yes	≥ 20	H	H	H	M	L	M	H	H	L
	Layer Fill	H	Yes	≥ 15	H	H	H	M	L	M	L	L	L
	Diked Containment	L	Yes-No	≥ 20	H-M	H	H	M	L	M	H	H	L
Residual/Refuse Co-Disposal	• Residual Spread With Refuse, Covered With Soil	H	No	≥ 3	L	H	H	M	M	M	M	M	L
	• Trench, Covered With Refuse and Soil	H	No	≥ 3	L	L	L	M	M	L	M	M	L
	• Residual Dumped on Refuse Covered With Soil	H	NO	≥ 3	L	H	H	M	M	L	M	M	L
	• Residual Dried in Piles, Spread With Refuse, Covered With Soil	H	NO	≥ 3	L	M	H	M	M	M	M	L	L
	• Residual Mixed With Soil, and Applied To Refuse		Yes	≥ 20	M-L	H	H	M	M	M	M	M	L

Key: H - High Potential (Comparatively)
M - Moderate Potential "
L - Low Potential "

TABLE 3. THE MAXIMUM POSSIBLE RESIDUAL/REFUSE MIX RATIOS FOR RESIDUALS OF VARIOUS SOLIDS CONTENT*

Residual Percent Solids Content	Maximum Residual/ Refuse Mix Ratio +
5	1/5
10	1/5
15	1/4
20	1/4
25	1/3
30	1/3

* Based on refuse absorption capacity of 0.6 lbs of water per lb of dry wt solid waste, and initial refuse moisture content of 65 percent

+ Weight ratio

The composition and strength of WTP residual leachates has been discussed. It is speculated that the various types of residuals can influence the rate of leachate generation through the effects of permeability. Residuals of low permeability can retard or reduce the flux of leachate from the fill; a more permeable residual may hasten leachate flux. Figure 3 qualitatively compares leachate flux among various types of residuals based on measured specific resistance* values. Coagulated raw and digested residuals are more permeable than non-coagulated raw, digested, and activated residuals. The coagulated residuals, being agglomerates of smaller particles, allow fluid to pass through more readily than do the non-coagulated residuals. The water retention characteristics of the non-coagulated fines could be a dominant factor in leachate production. Thus, coagulation processes which tend to increase the dewaterability of WTP residuals may present a disadvantage from the standpoint of leachate generation.

The specific gas production from anaerobic digestion of raw primary WTP residuals generally ranges between 12 to 17 ft^3 per lb of volatile solids destroyed. Of this gas, between 42.5 and 75 percent may represent methane (6). Complete microbial degradation of one pound of municipal refuse under optimal conditions yields approximately 4-5 ft^3 of methane (8,14). Potential methane production is therefore greater for raw primary residuals than for municipal refuse. Figure 4 presents the methane production potential for various WTP residuals based on the average percent volatile content of the residuals (5).

The stabilization of WTP residuals prior to landfilling is highly site-specific. Unstabilized residuals are usually unsuitable for landfilling because of obnoxious odors. A comparison of leachate constituent concentrations for raw and digested residuals revealed no apparent differences, though the data used was primarily from residual/refuse co-disposal landfill sites. The reduction in residual pathogen populations afforded by stabilization is important from a public health

TABLE 4. OPERATIONAL CONSTRAINTS OF RESIDUAL MOISTURE CONTENT

Percent Solids	Workability and Application
3 - 15	• Free flowing fluid co-disposal operations only
15 - 20	• Will not support cover material, i.e. soil sinks to the bottom of sludge. Applicable for narrow trenching due to soil bridging, and operations where soil bulking is provided.
20 - 28	• Solid content is high enough to support soil cover, though soils still have a tendency to sink into the sludge. Sludge is incapable of supporting equipment. Applicable for sludge only trenching, area-fill operations, or sludge/soil mixtures to act as final cover over completed landfills.
28 - 32	• Sludge capable of supporting equipment. Sludge still spreads out evenly in a trench when dropped from atop. Applicable for all sludge only operations
>32	• Sludge will not spread out evenly in a trench when dropped from atop. Applicable to area-fill mound and layer applications

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* Specific resistance can be defined as the pressure required to produce a unit rate of flow through a residual cake.

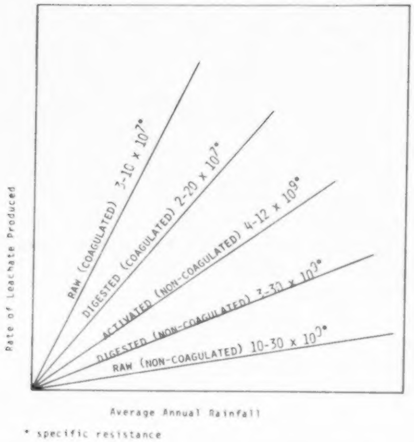

Figure 3. Relative rate of leachate producible based on residual permeabilities

standpoint.

Settlement due to volume reduction in landfilled residuals is important from an environmental as well as planning perspective. Settlement can promote cracking and create fissures in cover material, thus leading to an increase in potential odor and toxicant transmission and surface infiltration. Substantial verticle and/or horizontal displacement may occur which may influence future lift placement operations or land use. Residuals with low solids content (15-20%) can be expected to settle more than residuals with a higher solids content (>28%). Residuals may dewater through evaporation or infiltration (into porous soils), and as moisture is lost, compaction and settling results. Based on original and final void ratios, the estimated total consolidation settlement of WTP residuals of various solids content are listed in Figure 5 . From Figure 5, a residual of 8 ft depth and 25 percent solids content can settle 2.5 ft, representing a settlement of over 30 percent. Settlement rates as calculated closely match actual findings by Sikora (20).

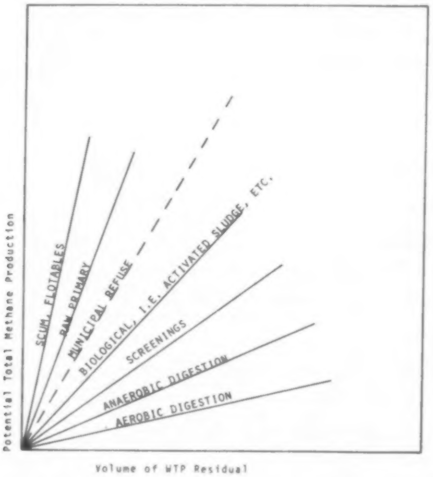

Figure 4. Comparison of WTP residuals and specific gas production

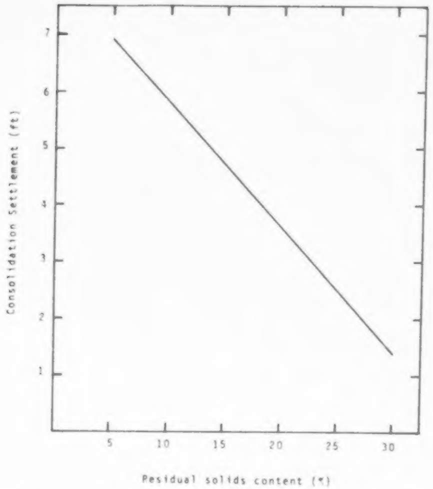

Figure 5. Consolidation settlement for residuals of different solids content

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LEACHATE AND GAS FROM MUNICIPAL SOLID WASTE LANDFILL SIMULATORS

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ABSTRACT

This study was initiated in 1974 to simulate the behavior of various types of landfills. The objective of the program was to determine the impact on solid waste decomposition from (1) varying moisture infiltration rates, (2) adding pH buffering compounds, (3) prewetting the wastes, (4) varying ambient temperatures, and (5) codisposing refuse with sewage sludge and various industrial wastes. To this end, a total of 19 landfill simulators were constructed in late 1974 and early 1975. Each was loaded with approximately 3.0 metric tons of municipal refuse. Small quantities of sewage sludge, industrial wastes, or other materials were added to 12 of these simulators. Maintenance, operation, and monitoring of these landfill simulators was then begun and has continued to this date. Water additions are made on all simulators on a prescribed schedule to simulate rainfall conditions in the Midwest U.S. Gas and leachate are collected, quantified, and analyzed. Refuse, soil, and ambient air temperatures are recorded.

This paper addresses data derived from the municipal-solid-waste-only test cells over an approximate six year period from November 1974 to August 1980. Moisture balances in each of these cells are quantified and evaluated. Six leachate analytical parameters have been selected, plotted, and scrutinized. Landfill gases are also examined with regard to their composition and generation rate.

INTRODUCTION

As directed by the Resource Conservation and Recovery Act of 1976 (RCRA), an inventory of all solid waste disposal sites across the nation is now being conducted by solid waste agencies in each of the 50 states. As part of the inventory process, state regulatory officials are required to both (1) identify all such disposal facilities in their state, and (2) evaluate these sites for compliance with EPA's "Criteria for the Classification of Solid Waste Disposal Facilities". These Criteria were published in the September 13, 1979 Federal Register, and require an assessment of the environmental adequacy of these facilities against each of eight criteria. Sites found to be in compliance with the Classification Criteria will be classified as "sanitary landfills" and likely allowed to continue operation. Sites found in non-compliance will be classified as "open dumps" and either closed immediately or allowed to upgrade their operations to that of compliance in accordance with a state-imposed upgrading schedule not to exceed five years.

Among the eight criteria, the one likely to cause the most serious compliance problem is the one dealing with ground water. For while promulgation of the RCRA Criteria is a large step forward in the regulatory area, the technical state-of-the-art for ground water protection undoubtedly requires additional work. It is generally conceded by most researchers that insufficient information is known in the areas of solid waste decomposition and contaminant generation/migration, the very areas that impact on ground water protection. If RCRA regulations in the ground water sector are to be enforced wisely, more will need to be known on these subjects in order to mandate appropriate upgrading actions against existing "open dumps", and to ensure appropriate design of future sanitary landfills.

This project was motivated by a need for such information. A broad objective of the program was to study solid waste decomposition and contaminant release at various types of landfills. Specific objectives are to determine:

1. The effect of different water infiltrations on solid waste decomposition.
2. The effect of sewage sludge additions on solid waste decomposition.
3. The effect of pH buffer addition on solid waste decomposition.
4. The effect of adding six selected industrial wastes on solid waste decomposition.
5. The effect of initial water addition on solid waste decomposition.
6. The survivability of polio-virus in landfills.
7. The effect of different air and surrounding soil temperatures on solid waste decomposition.
8. The ability to duplicate monitoring data from two test cells constructed and operated under similar conditions.

TEST CELL CONSTRUCTION

The project site for this study is the U.S. Environmental Protection Agency (EPA) Center Hill Laboratory in Cincinnati, Ohio. A total of 19 test cells were constructed for this program in November 1974 and April 1975. Fifteen of these test cells are located outdoors and below the ground surface. These 15 are arranged in a horseshoe alignment. The remaining four test cells are located inside the high-bay area of the Center Hill Laboratory facility, and are above-ground, located on a concrete slab. (See Figure 1)

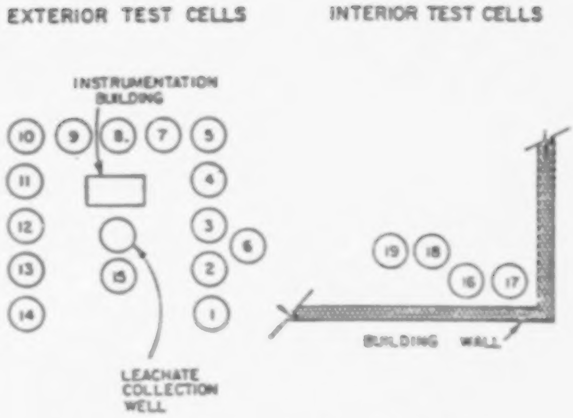

Figure 1. Test cell location plan

Individual test cells consist of steel tubes 1.83 m in diameter, 3.66 m high, and 4.76 mm thick. Steel sidewalls were coated with coal-tar epoxy as a moisture and gas seal. The outside test cells were placed on concrete slabs in an excavated area. Soil was then backfilled around the sidewalls of these 15 cells to within 0.3 m of the top of each steel tube. Several layers of fiberglass cloth were placed inside each cell covering the concrete base and extending 0.3 m up the sidewall to further ensure a water-tight and gas-tight seal. Interior test cells were placed atop steel bases which were welded onto the bottom of the test cell tubes.

Provisions for leachate drainage were installed in all cells. Exterior cells had a small depression in the concrete slab and connective piping to a leachate collection well, located in the center of the horseshoe layout of test cells. This well serves as a central leachate collection point for all exterior cells and is also used as a ground water drawdown to prevent pressure and infiltration of ground water into the test cells. Interior cells are mounted on concrete blocks and leachate drains attached below the cell bottoms.

All test cells were then readied for waste loading. To minimize the exposure time and, thus more closely simulate actual landfill conditions, all test cells were completely loaded and their tops welded shut within seven days of waste delivery. Initially, a 15.2 cm thick layer of silica gravel was applied at the bottom of each cell. This layer serves as a reservoir for leachate and prevents refuse from clogging the bottom drain. Silica gravel was selected for its inert nature to prevent any chemical reaction with the collected leachate.

Refuse was then delivered to the site and added to each test cell in eight 0.3 m thick increments or "lifts". Each lift was compacted with a wrecking ball. Sludges and other materials added to selected cells were applied at the top of each lift in proportionate amounts (except the first lift to avoid premature leaching). Temperature probes were installed atop the second, fourth, and sixth lifts in each cell. Gas probes were installed atop the second and sixth lifts.

Upon completion of waste loading, a 0.3 m thick layer of silty clay cover soil was applied atop the waste. An additional 0.3 m thick layer of pea gravel was placed atop the cover soil and a gas probe and water distribution ring installed in the gravel. Steel lids were then welded into place at the top of each test cell, providing an air- and water-tight seal. A cross section of a typical test cell is shown in Figure 2.

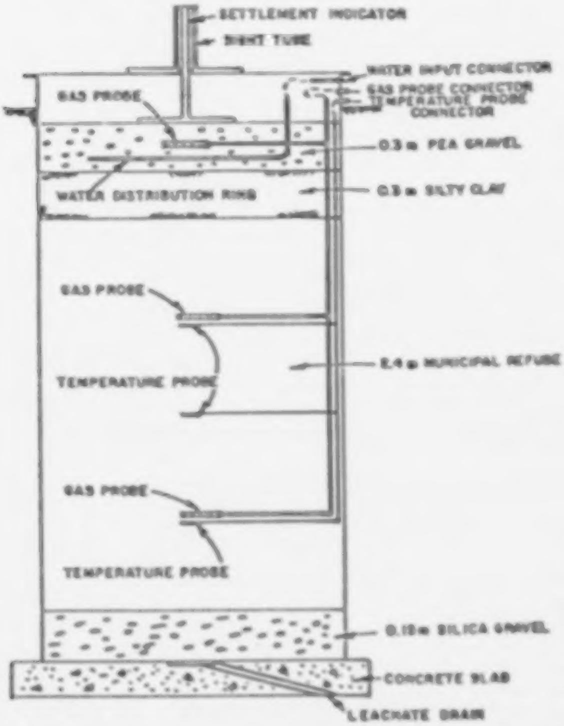

Figure 2. Cross-section of typical test cell.

A summary of the program plan covering all 19 test cells has been included as Table 1. This tabulation indicates the construction data, location, additive type, and annual infiltration rate for each of the 19 cells. As mentioned previously and indicated in Table 1, sludges, industrial waste, and other materials were added to selected cells. The intent of these additions was to allow an investigation of what effect codisposing these materials with municipal refuse would have on solid waste decomposition processes. More specifically, this arrangement provided test cells which fulfilled the earlier objectives as follows:

1. Different water infiltration rates: Test Cells 1, 2, 3, and 4.
2. Sewage sludge addition: Test Cells 5, 6, and 7.
3. pH buffer addition: Test Cell 8.
4. Six selected industrial sludge additions: Test Cells 9, 10, 12, 13, 14, and 17.
5. Initial water addition: Test Cell 11.
6. Survivability of polio-virus: Test Cell 15.
7. Different ambient air and soil temperatures: Test Cells 2 and 16.
8. Experimental replication: Test Cells 16, 18, and 19.

TABLE 1. DESIGN, CONSTRUCTION, AND OPERATIONAL FEATURES OF ALL TEST CELLS

Test Cell	Initial Loading Additive	Annual Infil-tration	Location	Construction/ Loading Date Nov. 1974	Construction/ Loading Date April 1975
1	--	203 mm	Outside	x	
2	--	406 mm	Outside	x	
3	--	607 mm	Outside	x	
4	--	813 mm	Outside	x	
5	Sewage Sludge*	406 mm	Outside	x	
6	Sewage Sludge*	406 mm	Outside	x	
7	Sewage Sludge*	406 mm	Outside	x	
8	Calcium Carbonate	406 mm	Outside	x	
9	Petroleum Sludge	406 mm	Outside		x
10	Battery Waste	406 mm	Outside		x
11	Water	406 mm	Outside	x	
12	Electroplating Waste	406 mm	Outside	x	
13	Inorganic Pigment Waste	406 mm	Outside	x	
14	Chlorine Prod. Brine Sludge	406 mm	Outside	x	
15	Polio-virus	406 mm	Outside	x	
16	--	406 mm	Inside	x	
17	Solvent Based Paint Sludge	406 mm	Inside	x	
18	--	406 mm	Inside		x
19	--	406 mm	Inside		x

* Low, medium, and high quantities of sewage sludge were added for Test Cells 5, 6, and 7, respectively.

Before loading, two representative samples were collected from each lift in each cell for purposes of waste characterization, chemical analysis, and moisture content determination. First, a total of 11 refuse categories were established and the waste characterization sample from each lift sorted in accordance with these categories. Subsequently, this data was averaged across all eight lifts in each cell to determine the waste composition of each test cell. Results for the solid-waste-only test cells investigated herein are presented in Table 2. These results indicate that the waste composition is relatively consistent from cell to cell, and in fact is truly representative of residential/commercial solid waste in the U.S. After completion of the sorting exercise for each lift, the waste from similar sort categories of all lifts and test cells were combined and a sample collected from each sort category for a chemical analysis. The results of this analysis are reflected in Table 3. Finally, a moisture content determination was performed on the sample from each test cell lift which was extracted for this purpose. The results of this determinationwere again averaged across each test cell, yielding some of the figures shown in Table 4.

As shown, moisture contents were found to range from 32 to 35 percent on the municipal-solid-waste-only test cells (Cells 2, 3, 4, 16, 18, and 19). Considering a measured in-place wet weight of approximately 3,000 kg, the dry weight of these cells was found to range from 1,946 to 2,042 kg. Considering a waste compartment size of 6.41 m^3, an initial refuse density can be calculated. On a wet weight basis, refuse densities were found to range between 466 and 470 kg/m^3. On a dry weight basis, densities ranged between 304 and 319 kg/m^3.

OPERATION AND MONITORING

To more closely simulate actual in-field landfill conditions, all test cells are operated in accordance with a strict monthly schedule of moisture application. This infiltration schedule has been summarized in Table 5. As shown, Test Cells 3 and 4 receive approximately 600 and 800 mm of annual infiltration, respectively. The remaining test cells (Cells 2, 16, 18, and 19) receive approximately 400 mm of annual infiltration.

In order to simulate actual precipitation/evaporation conditions in the Midwest U.S., this annual rate is administered on a periodic (monthly) basis, but in non-equal quantities. As indicated, no infiltration is applied in the usually dry months of January, July, August, and September. High amounts of infiltration are applied in the normally wet months of March, April, and May. Low-rate applications are made in the remaining transition months.

Physical data recording, sampling, and analysis are performed on all 19 test cells; a summary of these activities on the municipal-solid-waste-only cells has been included in Table 6. To summarize this schedule, gas volumes and refuse and air temperatures are determined on interior test cells each day. Refuse, soil, and air temperatures on outside test cells are determined bi-weekly. Gas composition is determined for both interior and exterior test cells monthly. Leachate samples are collected each month, their volumes recorded, and representative samples prepared. These samples are then analyzed for 22 parameters each month, with an additional 12 parameters each quarter and seven parameters every six months. Analyses are performed at the Center Hill

TABLE 2. REFUSE CHARACTERIZATION FOR SELECTED TEST CELLS [2]

	Percent Composition										
Test Cell	Food Waste	Garden Waste	Paper	Plastic, Rubber, Leather	Textiles	Wood	Metal	Glass	Ash, Rock, Dirt	Diapers	Fines
2	6.3	21.5	41.5	5.4	4.5	2.2	8.5	9.9	3.3	1.2	3.5
3	11.0	30.2	34.9	8.3	3.3	0.3	8.6	5.9	3.4	2.3	2.4
4	5.9	22.6	37.8	6.4	4.1	2.0	5.9	8.8	1.4	3.2	3.5
16	7.3	16.3	41.8	10.1	3.8	0.8	8.9	6.0	1.3	2.7	2.8
18	6.2	9.8	48.3	6.1	3.3	3.9	9.3	8.4	3.0	2.9	3.4
19	8.5	11.9	43.2	7.5	2.7	1.9	7.4	10.0	1.9	3.0	3.6
Average*	8.1	15.7	41.8	6.9	4.1	1.8	8.3	7.8	2.9	2.4	3.2
Std.Dev.*	2.2	5.3	4.5	1.8	1.5	1.0	1.4	1.4	1.3	1.0	0.7

* For all 19 test cells.

TABLE 3. REFUSE CHEMICAL ANALYSIS FOR ALL TEST CELLS [3]

Component	Paper	Garden	Metal	Glass	Food	Plastics, rubber, leather	Fines†	Ash, rocks, dirt	Diapers	Wood	Composite
COD§	0.804	0.815	0.492	0.011	0.754	2.16	0.935	0.040	0.720	0.503	0.520
TKN	0.028	0.171	0.022	0.140	3.09	1.25	0.131	0.119	0.138	0.228	0.267
Total Phosphate	0.048	3.14	2.79	0.049	10.4	1.40	1.97	4.48	2.65	0.103	2.32
Lipids	2.47	3.06	0.420	1.54	13.8	5.02	4.85	1.32	2.26	1.00	2.84
Ash	92.0	36.5	4.85	2.25	41.6	107	49.5	19.6	96.0	77.9	25.8
Crude fiber	21.7	16.6	0.235	0.060	10.5	21.5	6.76	5.85	13.7	20.8	11.3
Total carbon	58.0	14.4	4.80	0.750	19.5	15.8	16.4	13.4	64.5	51.0	24.8
Inorganic carbon	4.30	4.66	3.40	0.220	2.50	5.75	4.30	7.80	0.740	0.380	3.08
Organic carbon	53.7	9.74	1.40	0.530	17.0	10.1	12.1	5.60	63.8	50.7	21.8
Sugar as Sucrose	<0.1	1.71	<0.1	<0.1	6.08	<0.1	1.18	<0.1	<0.1	<0.1	3.20
Starch	3.40	7.42	<0.1	<0.1	8.57	3.42	7.20	6.40	<0.1	0.78	16.2
Asbestos	NA¶	NA	NA	NA	NA	NA	NA	NA	NA	NA	<1**
Arsenic††	<0.1	NA	<0.1	10.2	NA	NA	1.2	3.6	<0.1	<0.1	80
Selenium††	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	<0.1
Mercury††	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	5.0
Lead††	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	15.0
Beryllium††	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	<0.1
Cadmium††	0.36	NA	20.9	2.7	NA	1.0§§	4.2	4.5	0.25	1.6	2.4
Iron††	375	330	625*	3220	505	444	0.392*	0.340	99.0	0.378	0.782*
Zinc††	50.0	106	175	9.75	59.0	118	322	101	363	59.4	127
Chromium††	8.2	1.1	15.3	1.1	1.3	2.0	13.1	10.1	0.5	1.1	10.7
Manganese††	13.1	194	870	15.7	12.2	12.1	115	177	3.90	50.0	90.5
Potassium††	11.2	0.135*	1.00	2.70	0.162*	98.7	135	555	750	90.0	475
Magnesium††	160	4175	80.5	472	377	289	1.02*	2.63*	279	233	0.225*
Calcium††	77.5	0.830*	<0.25	16.2	0.465*	912	2.11*	4.08*	380	590	0.437*
Sodium††	9.70	185	37.0	60.0	804	143	600	0.209*	0.110*	372	950
Copper††	4.5	9.34	0.221*	2.54	8.58	12.4	35.8	32.6	4.14	38.2	31.6
Nickel††	15.7	13.7	115	19.0	12.5	32.0	33.2	10.1	3.36	27.0	10.1
Moisture	56.7	136.4	8.80	2.00	[illegible]	57.04	123	30.79	133	21.43	
Moisture¶¶	35.20	56.91	6.18	1.65	70.07	69.27	69.76	18.52	66.28	17.10	
Composition	42.6	10.7	12.2	12.2	3.6	8.7	2.9	3.2	1.3	2.6	
Composition¶¶	41.62	13.77	8.21	7.83	7.36	10.91	3.58	3.36	2.47	1.99	

* Percent by dry weight unless otherwise specified.

† <25.4 mm (1.0 inch).

§ g COD/g sample.

¶ Not analyzed.

** Fibers per gram.

†† Parts per million by weight.

§§ Plastics, rubber, and leather not analyzed.

¶¶ Percent wet weight.

TABLE 4. INITIAL LOADING FEATURES FOR SELECTED TEST CELLS

	Test Cell					
	2	3	4	16	18	19
Refuse Weights						
Wet weight (kg)	2,989	3,007	3,002	2,996	3,000	3,012
Dry weight (kg)	1,946	1,958	1,954	1,950	2,034	2,042
Initial Moisture						
Percent of wet weight (%)	35	35	35	35	32	32
Quantity (l)	1,043	1,049	1,048	1,046	966	970
Refuse Volume (m^3)	6.41	6.41	6.41	6.41	6.41	6.41
Initial Refuse Density						
Wet weight basis (kg/m^3)	466	469	468	467	468	470
Dry weight basis (kg/m^3)	304	305	305	304	317	319

TABLE 5. MONITORING PROGRAM FOR SELECTED TEST CELLS

Media	Determination	Frequency	Test Cell 2	3	4	16	18	19
Gas	Methane (CH_4)	Monthly	x	x	x	x	x	x
	Carbon Dioxide (CO_2)	Monthly	x	x	x	x	x	x
	Nitrogen (N_2)	Monthly	x	x	x	x	x	x
	Oxygen (O_2)	Monthly	x	x	x	x	x	x
	Hydrogen (H_2)	Monthly	x	x	x	x	x	x
	Volume	Daily				x	x	x
Refuse/Soil/Air	Temperature	Bi-weekly	x	x	x			
	Air Pressure	Bi-weekly	x	x	x			
	Temperature	Daily				x	x	x
	Air Pressure	Daily				x	x	x
Leachate	Volume	Monthly	x	x	x	x	x	x
	Temperature	Monthly	x	x	x	x	x	x
	Total Organic Carbon (TOC)	Monthly	x	x	x	x	x	x
	Chemical Oxygen Demand (COD)	Monthly	x	x	x	x	x	x
	Total Solids (TS)	Monthly	x	x	x	x	x	x
	Total Volatile Solids (TVS)	Monthly	x	x	x	x	x	x
	pH	Monthly	x	x	x	x	x	x
	Oxidation Reduction Potential (ORP)	Monthly	x	x	x	x	x	x
	Specific Conductance	Monthly	x	x	x	x	x	x
	Alkalinity	Monthly	x	x	x	x	x	x
	Total Kjeldahl Nitrogen (TKN)	Monthly	x	x	x	x	x	x
	Ortho-phosphate (Ortho-PO_4)	Monthly	x	x	x	x	x	x
	Iron (Fe)	Monthly	x	x	x	x	x	x
	Cadmium (Cd)	Monthly	x	x	x	x	x	x
	Sulfur (S)	Monthly	x	x	x	x	x	x
	Chloride (Cl)	Monthly	x	x	x	x	x	x
	Mercury (Hg)	Monthly	x	x	x	x	x	x
	Vanadium (V)	Monthly	x	x	x	x	x	x
	Beryllium (Be)	Monthly	x		x			
	Selenium (Se)	Monthly	x		x			
	Cyanide (CN)	Monthly	x		x			
	Phenol	Monthly	x		x			
	Arsenic (As)	Monthly	x		x			
	Hexavalent Chromium (Cr^6)	Monthly	x		x			
	Copper (Cu)	Quarterly	x	x	x	x	x	x
	Lead (Pb)	Quarterly	x	x	x	x	x	x
	Nickel (Ni)	Quarterly	x	x	x	x	x	x
	Zinc (Zn)	Quarterly	x	x	x	x	x	x
	Asbestos	Quarterly	x		x			
	Tin (Sn)	Quarterly	x	x	x	x	x	x
	Total Phosphate (Total PO_4)	Quarterly	x	x	x	x	x	x
	Organic Acids C_2 to C_6	Quarterly	x	x	x	x	x	x
	Fecal Coliform	Quarterly	x	x	x	x	x	x
	Fecal Streptococci	Quarterly	x	x	x	x	x	x
	Total Chromium (Total Cr)	Quarterly	x	x	x	x	x	x
	Arsenic (As)	Semi-annually	x	x	x	x	x	x
	Phenol	Semi-annually	x	x	x	x	x	x
	Beryllium (Be)	Semi-annually	x	x	x	x	x	x
	Titanium (Ti)	Semi-annually	x	x	x	x	x	x
	Boron (B)	Semi-annually	x	x	x	x	x	x
	Vanadium (V)	Semi-annually	x	x	x	x	x	x
	Antimony (Sb)	Semi-annually	x	x	x	x	x	x

TABLE 6. INFILTRATION SCHEDULE FOR SELECTED TEST CELLS

Month	Quantity Per Month (liters) 406 mm Annual Rate*	607 mm Annual Rate+	813 mm Annual Rate#	Types Per Month
January	0	0	0	None
February	96.90	145.72	194.17	Low
March	194.17	291.45	388.34	High
April	194.17	291.45	388.34	High
May	194.17	391.45	388.34	High
June	96.90	145.72	194.17	Low
July	0	0	0	None
August	0	0	0	None
September	0	0	0	None
October	96.90	145.72	194.17	Low
November	96.90	145.72	194.17	Low
December	96.90	145.72	194.17	Low
Total	1,067.01	1,602.95	2,135.87	

* Test Cells 2, 16, 18, and 19.
+ Test Cell 3.
# Test Cell 4.

Laboratory using facilities and equipment assigned specifically to this project. Quality assurance program procedures are utilized to ensure the validity of sampling results. Analytical data derived from the project is punched onto computer cards and entered into the memory banks of an EPA in-house computer. This computational facility can then be used for plotting project data as an aid in evaluating results.

MOISTURE BALANCE EVALUATION

As stated previously, leachate and gas data and evaluations in this paper will concentrate on those cells designated as "municipal-solid-waste-only" (i.e., cells having received no waste additives). The first task then will be to evaluate the relation between moisture added and moisture leached in each of the six subject test cells. Data on moisture added and leached plotted as "interative liters of water per kilogram of solid waste versus time" is presented in Figures 3 through 8. As indicated, Test Cells 2, 16, 18, and 19 all receive approximately 400 mm of infiltration per year. Test Cells 3 and 4 receive 600 and 800 mm per year, and thus their plots are proportionally higher.

Several observations can be made from the data presented in these figures. First, it is obvious that infiltration can be added for a period of time without significant quantities of leachate being generated. Second, although leachate volumes increase and decrease from month to month in general accordance with the variation in infiltration volumes added, leachate volume changes appear to lag corresponding infiltration changes by about one to two months. Third, even after the onset of leachate generation in measurable quantities, infiltration continues to be added each month at quantities in excess of subsequent leachate volumes.

Infiltration and leachate volume data on a cumulative basis is presented for Test Cells 2, 3, 4, 16, 18, and 19 in Figures 9 through 14, respectively. The upper plot on each of these respective curves reflects in liters/kilogram the sum of cumulative moisture added (via infiltration) plus the initial moisture content (i.e., inherent in the waste as delivered). The lower plot in each of these figures reflects the cumulative quantity of moisture leached. Again it is obvious that the cumulative amount of moisture leached is considerably less than the moisture added for any point in time.

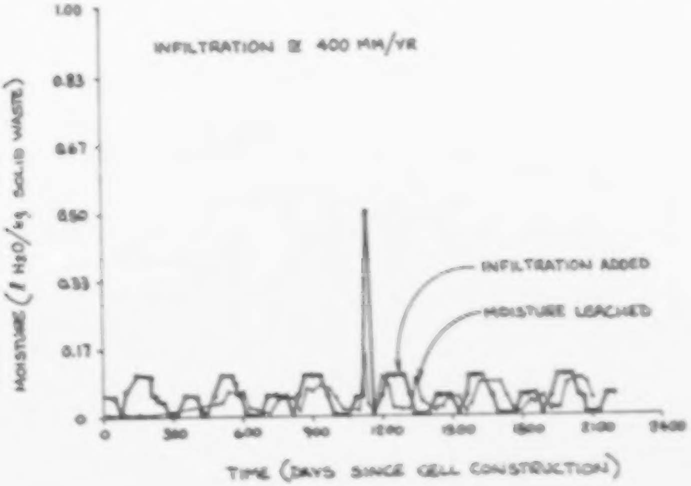

Figure 3. Moisture added/leached at Test Cell 2.

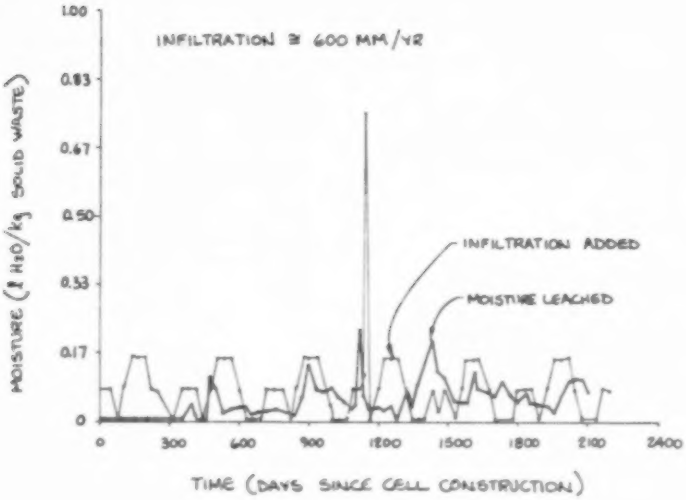

Figure 4. Moisture added/leached at Test Cell 3.

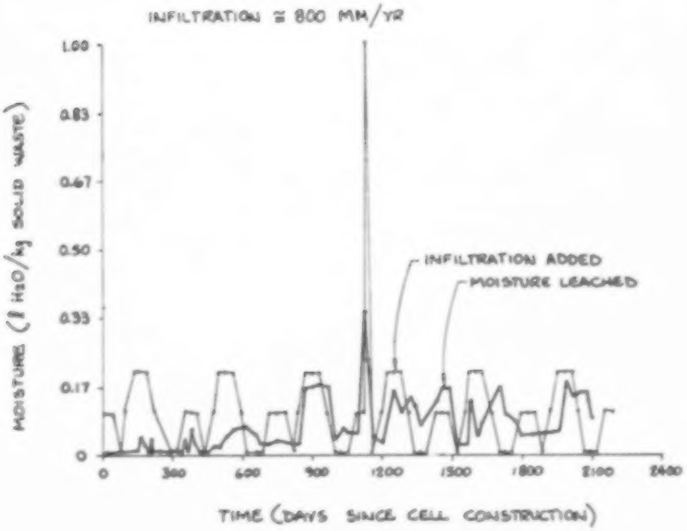

Figure 5. Moisture added/leached at Test Cell 4.

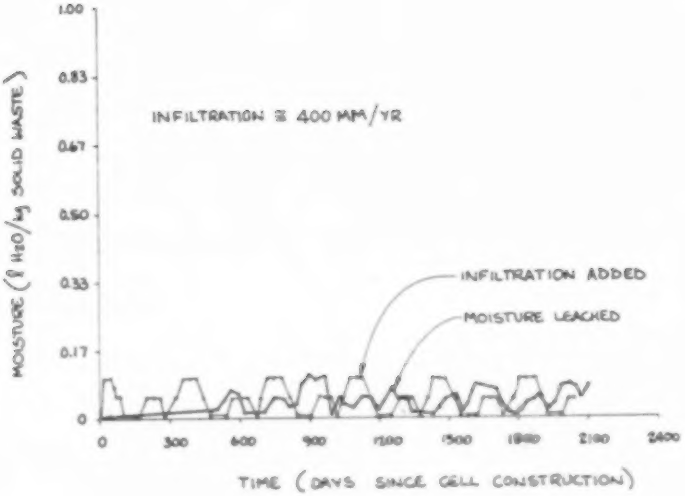

Figure 6. Moisture added/leached at Test Cell 16.

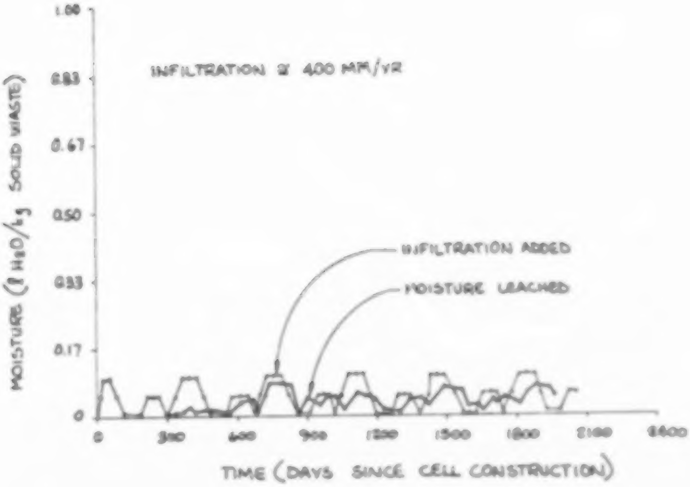

Figure 7. Moisture added/leached at Test Cell 18.

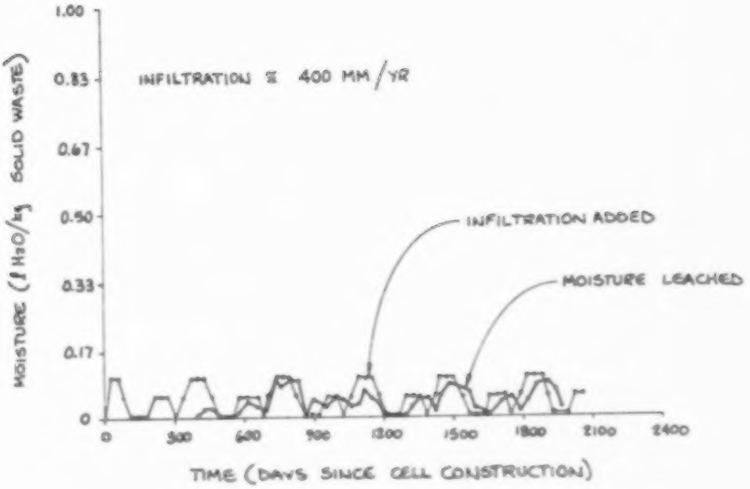

Figure 8. Moisture added/leached at Test Cell 19.

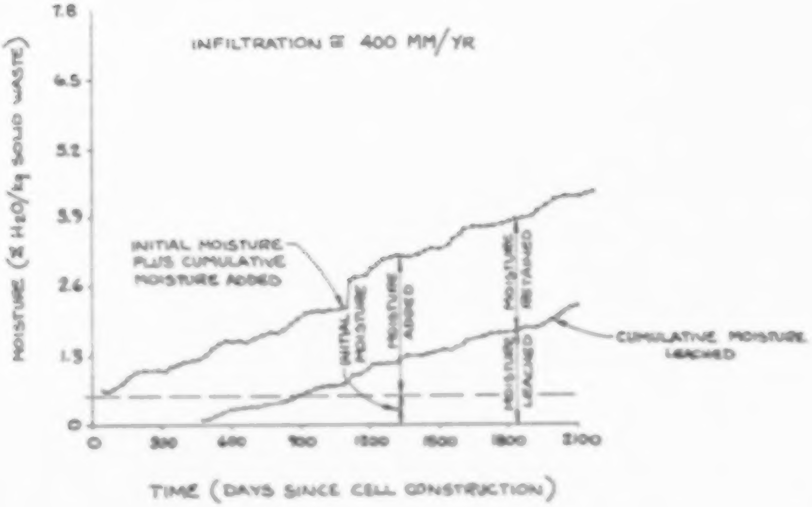

Figure 9. Moisture balance for Test Cell 2.

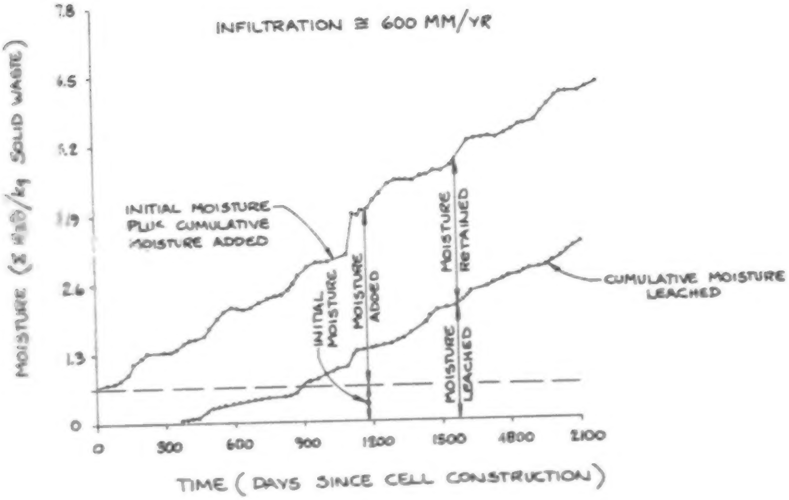

Figure 10. Moisture balance for Test Cell 3.

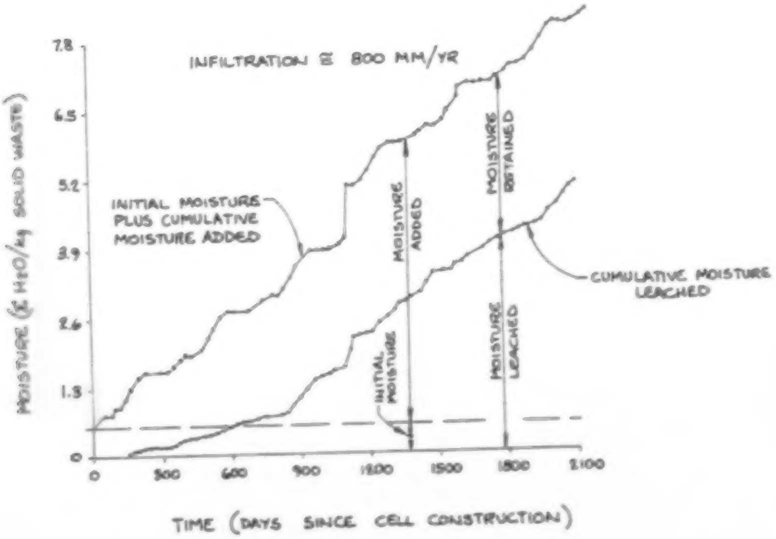

Figure 11. Moisture balance for Test Cell 4.

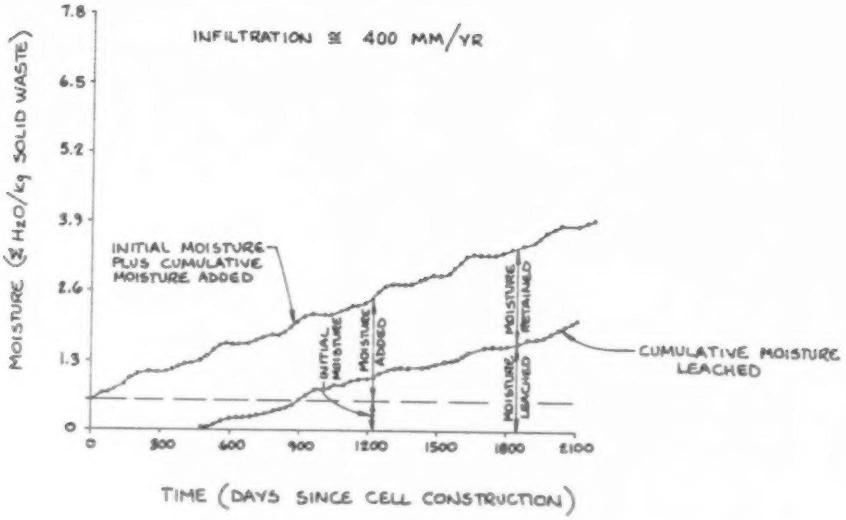

Figure 12. Moisture balance for Test Cell 16.

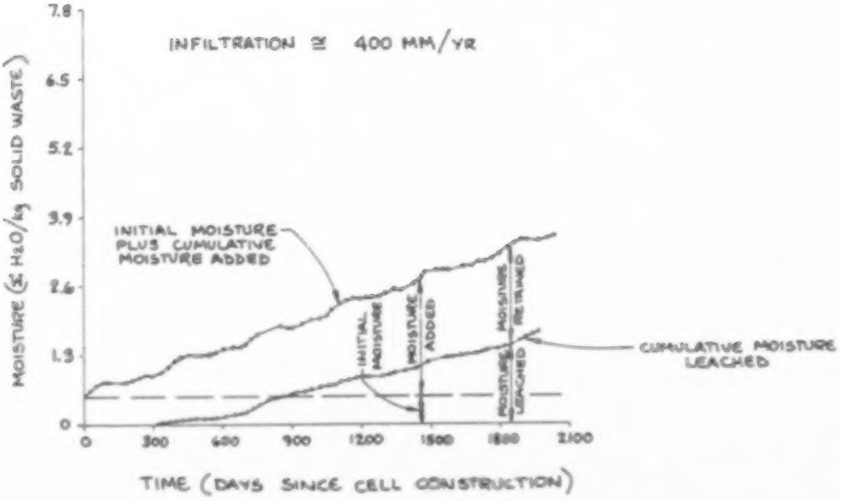

Figure 13. Moisture balance for Test Cell 18.

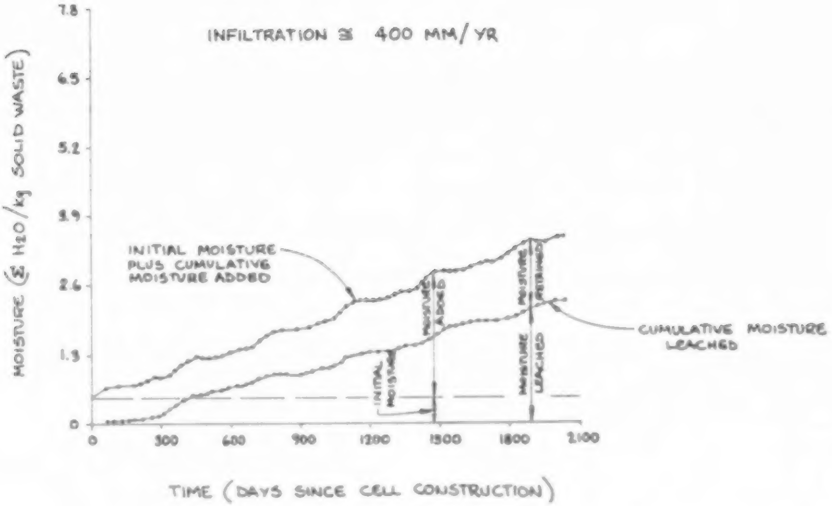

Figure 14. Moisture balance for Test Cell 19.

This, of course, is due to the ability of refuse to retain a certain quantity of moisture. Generally, if the initial moisture of the refuse plus the moisture added does not exceed this retention ability, leachate will not appear. This quantity is known as the field capacity and is technically defined as "the maximum moisture content which a soil (or solid waste) can retain in a gravitational field without producing continuous downward percolation". [4] Generally, the release of significant quantities of leachate marks the initiation of this continuous leaching and the point at which field capacity is attained. Of course, some leachate may be emitted before this point; but generally these are small quantities and can be followed by times at which no leachate is emitted despite the addition of infiltration water.

As shown in Figures 9 through 14, field capacities were attained at times ranging from 339 to 540 days for the 400 mm/year test cells (i.e., Cells 2, 16, 18, and 19). Although it receives a higher infiltration rate of 600 mm/year, Test Cell 3 also reached field capacity within this time range at 480 days. Test Cell 4 initiated leaching at 150 days, more appropriate considering its much higher infiltration rate of 800 mm/year.

Moisture conditions present in each of the above described test cells at the onset of continuous leaching are detailed in Table 7. The equation used to define the water balance in each cell at that time is:

Moisture Retained = (Initial Moisture) + (Moisture Added) - (Moisture Leached)

As indicated in Table 7, the moisture retained in each cell at the point of initial continuous leaching (i.e., attainment of field capacity) was found to exist within a fairly tight range of 1.02 to 1.31 l/kg for all but Test Cell 18 which appears to have initially achieved field capacity at 0.85 l/kg. (Note that all mass values from this point on are on a dry-weight basis.) From this data it is apparent that attainment of field capacity is dependent upon the moisture retained (expressed as liters of moisture retained per kilogram of dry solid waste); the time it takes to reach field capacity does not appear to have any direct relation to either the moisture application rate or initial moisture content by themselves.

Presumably, after field capacity has been reached, one liter of moisture should be leached for each one liter of moisture added. Thus, the moisture retained should stabilize after the onset of continuous

TABLE 7. MOISTURE BALANCE SUMMARY

Test Cell	Annual Infiltration mm	Annual Infiltration l/kg	Moisture to Continuous Leaching No. of Days	Moisture to Continuous Leaching Initial Moisture (l/kg)	Moisture to Continuous Leaching Moisture Added (l/kg)	Moisture to Continuous Leaching Moisture Leached (l/kg)	Moisture to Continuous Leaching Moisture Retained (l/kg)	Moisture to Date No. of Days	Moisture to Date Initial Moisture (l/kg)	Moisture to Date Moisture Added (l/kg)	Moisture to Date Moisture Leached (l/kg)	Moisture to Date Moisture Retained (l/kg)
2	406	0.55	540	0.54	0.49	0.01	1.02	2,099	0.54	3.69	2.32	1.91
3	607	0.82	480	0.54	0.80	0.03	1.31	2,099	0.54	5.52	3.36	2.70
4	813	1.09	150	0.54	0.70	0.05	1.19	2,099	0.54	7.37	5.39	2.52
16	406	1.10	476	0.54	0.65	0.01	1.18	2,099	0.54	3.24	2.13	1.65
18	406	1.05	339	0.47	0.38	0.00	0.85	1,962	0.47	2.87	1.80	1.54
19	406	1.05	395	0.48	0.62	0.00	1.10	1,962	0.48	2.85	1.90	1.43

leaching. However, this has not been the case. As shown in Figures 9 through 14, the gap between "initial moisture plus cumulative moisture added" and "cumulative moisture leached" has stabilized somewhat with the onset of initial continuous leaching. However, the size of this gap (or the moisture retained) appears to be growing slightly larger with time. This gap is quantified in Table 6. As shown, the six solid-waste-only test cells now retain between 1.43 and 2.70 l/kg. These figures demonstrate a substantial increase for Test Cells 3 and 4. While the moisture retained in Test Cells 2, 16, 18, and 19 remains below 2.00 l/kg, measurable increases in the moisture retained have also occurred in these cells.

The reason for these increases cannot be fully explained. One possible reason may be the effect of channeling. Channeling exists when moisture added to a waste percolates directly through it, failing to fill all the voids before leaching. Some voids remaining in the waste at the onset of continuous leaching may now gradually be filling with moisture, rather than letting the infiltration water channel directly through. Leaking test cell drains for the exterior lysimeters is another possible explanation, particularly for Test Cells 3 and 4 considering their high retained moisture figures. While no drain line leaks have been witnessed for these two test cells in the past two years, visible leaks have existed for some of the other test cells in this program. Because Cells 1 to 15 are all located outside, below the ground surface, some leaking is possible for all test cells, and it might go undetected. The effect of such leaking would allow leachate quantities to escape unmeasured, decreasing the measured quantity of moisture leached, and thus increasing the measured quantity of moisture retained to a figure higher than is actually the case.

The above explanation may be plausible for Test Cells 3 and 4. However, no leaks have ever been observed for the interior Test Cells 16, 18, and 19. Increased moisture retentions for these test cells thus may have to be explained by the channeling theory described above. Alternatively, it has been theorized that some moisture may be escaping as condensate in daily gas volumes collected and vented. However, an investigation was recently conducted to quantify the moisture content in the gas and to determine its impact on the overall moisture balance. This impact was determined to be insignificant at less than one percent.

LEACHATE QUALITY EVALUATION

Of the numerous leachate analytical parameters determined on this study, five have been selected for indepth investigation in this presentation. In addition, only data from Test Cell 2 (400 mm/year infiltration), Test Cell 3 (600 mm/year infiltration), and Test Cell 4 (800 mm/year infiltration) will be addressed in this section. Test Cells 16, 18, and 19 all receive the same amount of infiltration and it was the intent of this section to develop trends as related to varying infiltration rates.

It is appropriate to note that graphs included herein are plots of mass leached (cumulative kilograms of contaminant per kilogram of dry solid waste) as a function of leachate volume (cumulative liters of leachate/kilogram of dry solid waste). Plots of mass leached vs time were considered, since these might be more understandable to the designer. However, histograms collected to this end were observed to be somewhat irregular due to the seasonal fluctuations in applied moisture. As a result, trends demonstrated in the ensuing leachate plots were not as readily discernable on the histograms. The effect of plotting contaminant masses leached against cumulative leachate volumes has the effect of smoothing out these fluctuations, normalizing them for irregularities in applied moisture. To make the plots in this section more useful as a design aid, it is suggested that the reader consult Figures 9 through 14 to determine the expected leachate volume for a given infiltration rate and time. Once expected leachate volume is computed, contaminant release can be determined from the ensuing plots in this section.

Total Organic Carbon

The release of TOC appears to be approaching an asymptote (Figure 15) at which only nominal amounts of TOC will subsequently be released. This trend is most apparent for Test Cell 4 (800 mm/year infiltration) which has yielded the greatest cumulative leachate volume to date. The curve for the 400 mm/year test cell exhibits a slightly higher trend and thus, greater TOC released for a given leachate volume. This tendency for a greater TOC release rate with a lower infiltration rate is not supported by data from the 600 mm/year test cell however, as the mass release trending for this test cell is the lowest of the three.

Despite these differences, the overall trending of these three TOC curves is quite similar, with the cumulative TOC leached for a cumulative leachate volume being somewhat the same regardless of the infiltration rate. Thus, the effect of a higher infiltration rate is merely to accelerate the release process with time. Ultimately, it would appear, each cell will have similar overall TOC release. As shown in Table 3, the TOC in the solid waste when loaded in the test cells was determined to be about 22 percent on a dry-weight basis. Since approximately 40 g/kg of TOC has been leached by the 800 mm/year test cell to date, roughly 19 percent of the initial TOC has been leached by the 800 mm/year cell to this point in the project. This percentage leached is illustrated on the right vertical axis in Figure 15. Since the overall trend of the TOC curve appears to be asymptotic, it could be presumed that ultimate TOC leaching would not be much larger than this 19 percent value. Much of the balance of the initial TOC may remain in the solid waste, never made available for leaching. Of course, it should be pointed out that not all organic

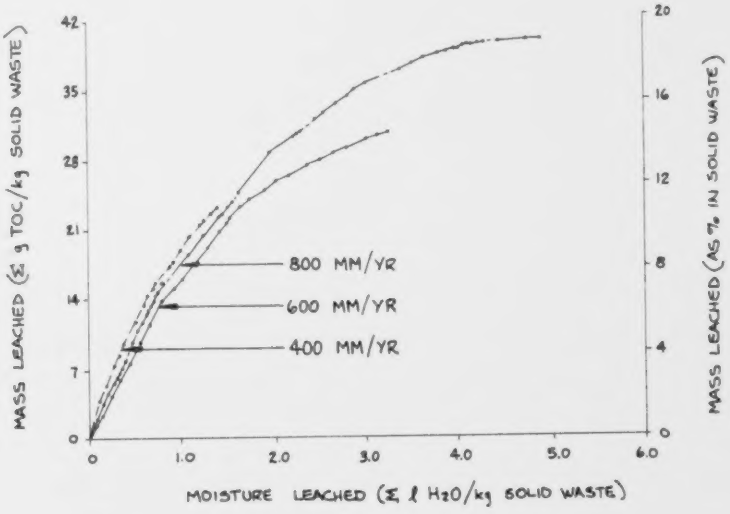

Figure 15. Mass leached: TOC.

carbon present in the waste at the time of loading is released in a liquid form; some of the organic carbon can be converted via microorganisms into gases such as carbon dioxide and methane.

A summary of masses leached to date (for TOC as well as other contaminants) has been included in Table 8. As shown, the 800, 600, and 400 mm/year cells have leached a cumulative total of 40, 30, and 23 g/kg (respectively) to date. These figures represented about 19, 14, and 11 percent of the TOC content found to exist in these cells at the time of initial loading.

A summary of concentrations for the various leachate contaminants addressed in this paper has been included in Table 9. As indicated, TOC concentrations attained maximum levels of between 31,000 and 37,000 mg/l early in the project. Since that time, TOC concentrations have gradually decreased to levels now between 1,000 and 2,000 mg/l. TOC concentrations are currently as low as 1,000 mg/l only for the 800 mm/year test cell, indicating that it has matured faster than the two lower infiltration rate cells, both having TOC concentrations no lower than 2,000 mg/l.

Chemical Oxygen Demand

COD removal (Figure 16) for the 400 mm/year cell exhibits the highest trajectory, although it correlates very closely to the trajectory for the 800 mm/year cell. The trend for the 600 mm/year cell is again lower than these two other cells and at times veers substantially off the trajectory established by the 800 and 400 mm/year cells.

Nonetheless, all three cells generally follow a similar asymptotic trending, and the slope for each of the curves can be said to be similar to that found for the TOC curve. In general, however, it appears that the COD curve is still climbing at a faster rate than that for TOC. This is to be expected since initially TOC is one of the primary contributors to the COD determination. However, inorganics also contribute to COD and as TOC tapers off in the future, inorganics may become the driving force behind COD in the leachate. The cyclical nature of the leachate pH (as indicated in plots contained in earlier papers on this project) is especially conducive to the continued appearance of COD in leachate. When pH values dip low and carbon dioxide is available in the absence

TABLE 8. SUMMARY OF CONTAMINANT MASSES LEACHED

	Mass Leached to Date (cumulative g/kg of solid waste)			Mass Leached to Date (as % in initial solid waste)		
Parameter	400 mm/ year*	600 mm/ year+	800 mm/ year#	400 mm/ year*	600 mm/ year+	800 mm/ year#
TOC	23	30	40	11%	14%	19%
COD	73	80	95	N/A	N/A	N/A
TKN	1.9	1.7	2.3	76%	68%	90%
TS	56	58	57	6%	7%	8%
Fe	0.8	1.3	1.4	9%	16%	17%
Ni	0.0013	0.0013	0.0019	12%	12%	18%

* Test Cell 2.
+ Test Cell 3.
# Test Cell 4.

N/A = Not applicable.

TABLE 9. SUMMARY OF CONTAMINANT CONCENTRATIONS IN LEACHATE

	Maximum Approximate Concentration to Date (mg/l)				Minimum Approximate Concentration to Date (mg/l)			
Parameter	400 mm/ year*	600 mm/ year+	800 mm/ year#	Overall **	400 mm/ year*	600 mm/ year+	800 mm/ year#	Overall **
TOC	31,000	37,000	32,000	37,000	2,000	2,000	1,000	1,000
COD	78,000	64,000	73,000	76,000	13,000	4,000	1,000	1,000
TKN	2,400	1,600	1,700	2,400	200	200	100	100
TS	64,000	46,000	56,000	64,000	7,000	7,000	3,000	3,000
Fe	650	1,000	500	1,000	200	50	100	50
Ni	2.6	2.0	1.9	2.6	0.1	0.1	0.1	0.1

* Test Cell 2.
+ Test Cell 3.
# Test Cell 4.
** Extremes for three test cells combined.

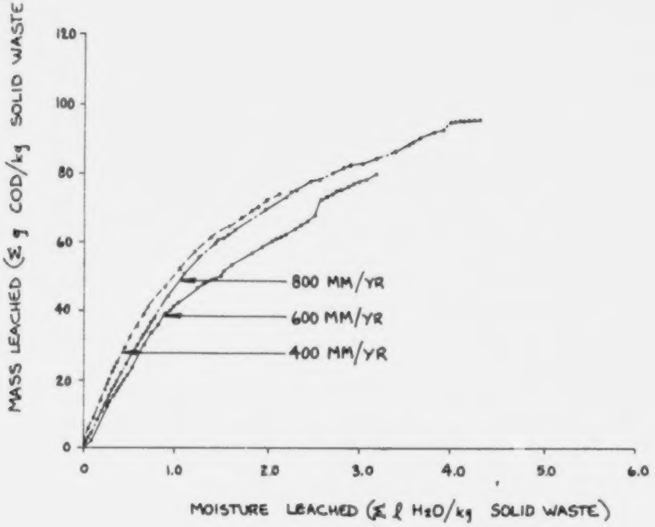

Figure 16. Mass leached: COD.

of oxygen, ferrous iron and other similar inorganic contributors to COD can be created and held in solution, soon to be manifested in the leachate.

Table 8 indicates that the 800, 600, and 400 mm/year cells have leached a total COD mass of about 95, 80, and 73 g/kg of solid waste to date. COD concentrations (Table 9) initially ranged between 31,000 and 37,000 mg/l for the three cells, but have gradually decreased to a range of 1,000 to 2,000 mg/l. Again, recent COD concentrations reflect the maturity of each cell's decomposition process, with current COD levels running at 1,000, 4,000, and 13,000 mg/l for the 800, 600, and 400 mm/year cells, respectively.

Total Kjeldahl Nitrogen

TKN leached by Test Cells 2, 3, and 4 has been plotted in Figure 17. Again the release exhibits an asymptotic trending for all three infiltration rates. The TKN plots for different cells cross paths for the first time; nonetheless, the 400 mm/year cell continues to have the highest overall trending, again followed in order by the 800 and 600 mm/year test cells.

Overall the deviation among the three cells appears to be greater than that found for TOC or COD. In addition, as shown on the right axis of this plot, from 68 to 90 percent of the TKN has been calculated to have been released in the leachate to date. This value is much higher than that seen in the release of other parameters, and is expected to be so since conditions are ideal early in solid waste decomposition for biological denitrification as a result of high temperature, acidic pH, and the presence of organics. Of course, whether up to 90 percent has actually been released is highly questionable. It is possible that the initial TKN reading in the solid waste is erroneous due to faulty analysis or compilation of a solid waste sample that was not truly representative of the waste mass at large.

As per Table 8, a total of 2.3, 1.7, and 1.9 grams of TKN per kg of solid waste has been leached by the 800, 600, and 400 mm/year test cells. Thus, for the first time a greater contaminant mass has been leached by a test cell receiving less infiltration (i.e., the 400 mm/year cell has emitted 1.9 g/kg versus only 1.7 g/kg for the 600 mm/year test cell). Since the 400 mm/year cell has emitted less leachate to date, obviously TKN concentrations in its leachate must be relatively high. This is demonstrated by the TKN concentrations reflected in Table 9. As indicated, initial concentrations of TKN were as high as 2,400 mg/l for the 400 mm/year cell,

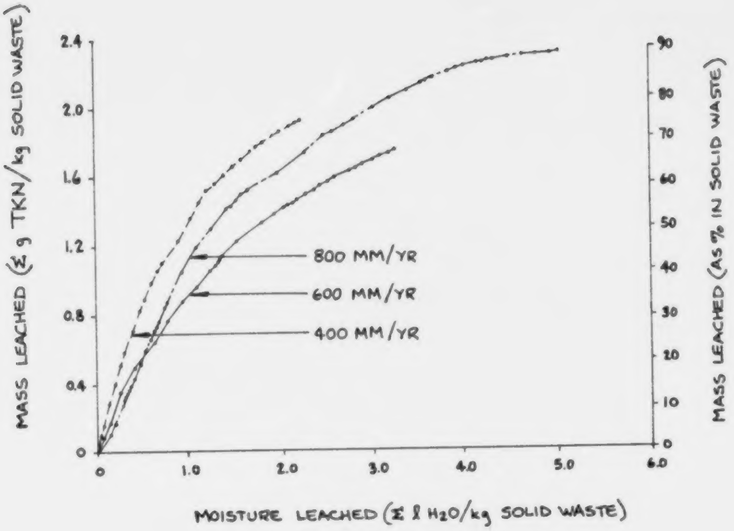

Figure 17. Mass leached: TKN.

versus only 1,600 and 1,700 mg/l for the 600 and 800 mm/year cells. Currently, however, TKN concentrations in leachate from the 400 mm/year cell have returned to the range of TKN concentrations from other cells (i.e., 200 mg/l versus 200 and 100 mg/l for the 600 and 800 mm/year cells).

Total Solids

Total solids released (Figure 18) also reveals an asymptotic trending. As for all previous plots, the 400 mm/year test cell is the highest followed in order by the 800 and 600 mm/year cells. The curve for the 400 mm/year test cell revealed a slightly greater divergence, with the 800 and 400 mm/year cells showing much closer correlation. Nevertheless, the overall trend for all three cells is similar; the asymptotic trending of these curves indicate that solids released is decreasing with time. As shown previously in Table 4, approximately 2,000 kg of dry solid waste were placed in each of the three cells being investigated herein. Thus, for the

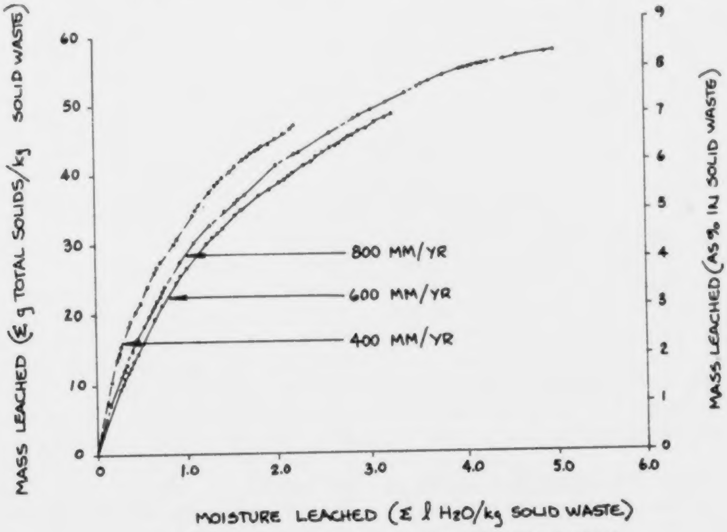

Figure 18. Mass leached: Total Solids.

800 mm/year test cell, approximately 8 percent of the available solids in that test cell have been leached to date. The asymptotic trending of the curve would indicate that ultimate release of solids with time would not be much greater than this amount.

Table 8 indicates that a total of 57, 48, and 46 g of total solids have been leached for each kilogram of dry solid waste for the 800, 600, and 400 mm/year cells. These quantities represent about 8, 7, and 6 percent respectively of the initial dry weight of solid waste loaded into the test cells. Total solids concentrations (Table 9) have gradually decreased with time. Initial solids concentrations ranged between 46,000 and 64,000 mg/l. Although the total solids leached to date was demonstrated to be greater for the 600 mm/year cell than the 400 mm/year cell, these figures were shown to be quite close. This fact may be reflected in the much higher initial concentrations of total solids as indicated in Table 9 for the 400 mm/year cell. Currently, total solids concentrations range between 3,000 and 7,000 mg/l. The mature stage of solid waste decomposition for the 800 mm/year cell is reflected by its solids concentration being in the lower part of this range.

Iron

Unlike previous parameters which have exhibited an asymptotic curve for mass released, iron release appears to be somewhat linear with time (Figure 19). As per previous curves, the 400 mm/year cell retains the highest trajectory. This time, however, the 800 mm/year cell retains the lowest trajectory (unlike the middle trajectory it held in previous plots). Lastly, although the three plots appear to correlate well over the first half of the monitoring period, recent trends have shown a greater divergence between the 600 and 800 mm/year cells. In fact, total iron mass released by the 600 mm/year cell is only slightly less than that for the 800 mm/year cell despite the fact that its leachate volume is only about two-thirds of that for the 800 mm/year cell.

As demonstrated in Table 8, total mass leached for these three cells is 1.4, 1.3, and 0.8 for the 800, 600, and 400 mm/year test cells. Based on the determination of iron content in the solid waste made at the onset of the project, these quantities amount to 17, 16, and 9 percent of the total iron in the wastes at that time. Due to the linear trending of all three plots, however, it would appear that iron will continue to leach at concentrations found

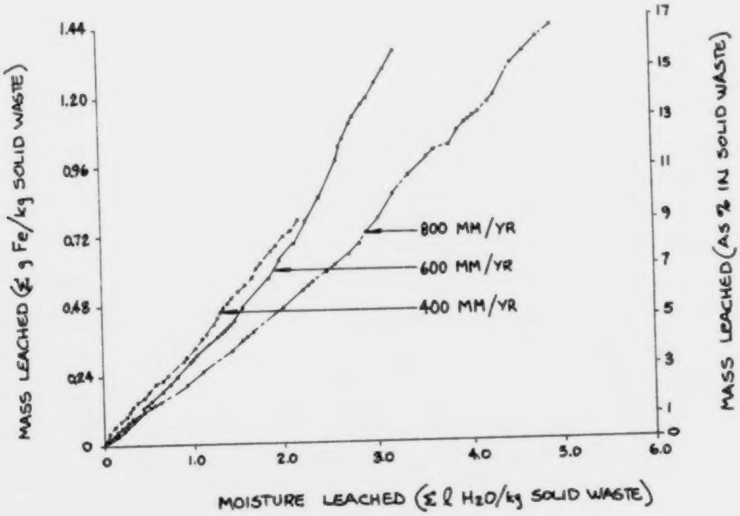

Figure 19. Mass leached: Iron.

in the past. Thus, it would appear that the total iron mass leached will ultimately run much higher than the 17 percent found to date. Maximum iron concentrations in leachate (Table 9) have ranged between 500 and 1,000 mg/l for the three test cells. The 1,000 mg/l concentration found for iron in the 600 mm/year cell is much higher than the maximum levels encountered in the other two cells and may be a partial explanation for its relatively high trajectory. Minimum iron concentrations in the leachate from these cells have ranged from 50 to 200 mg/l. Unlike for previous parameters it should be noted that there has been no gradual decrease in iron concentration with time.

Nickel

Mass leached for nickel is demonstrated in Figure 20. As for most of the previous plots, an asymptotic trending is observed. Again, the high, middle, and low trajectories represented the 400, 800, and 600 mm/year cells, respectively. As can be seen in Figure 20, recent trends have caused the 400 mm/year test cell to dip below the trajectory for the 800 mm/year test cell.

As shown in Table 8, nickel mass leached to date has ranged from 0.0013 to 0.0019 g/kg of solid waste. The total nickel mass leached for the 400 and 600 mm/year cells appears to be essentially the same despite that the 400 mm/year cell has leached only two-thirds the volume leached by the 600 mm/year cell. To date, 18 percent of the nickel found in the initial solid waste has been leached by the 800 mm/year cell, while about 12 percent has been leached for each of the 400 and 600 mm/year cells.

As would be expected from an asymptotic trending, nickel concentrations have gradually decreased with time. As reflected in Table 9, early concentrations ranged from 1.9 to 2.6 mg/l. A 2.6 mg/l maximum occurred in the 400 mm/year cell and may be a partial explanation for its initially high trajectory on a mass leached basis (Figure 20). However, nickel levels are now relatively low and essentially the same for all three test cells at about 0.1 mg/l.

TEMPERATURE EVALUATION

As stated previously, refuse temperatures are recorded on a regular basis for all test cells. Temperature plots for two selected cells have been included in Figure 21. Note that Test Cell 2 is an exterior cell placed below the ground surface, while Test Cell 19 is an interior cell placed above ground and maintained at room temperature.

Results for both cells demonstrate increased temperatures soon after construction. This trend is more obvious for the interior test cell with its highest temperatures reaching

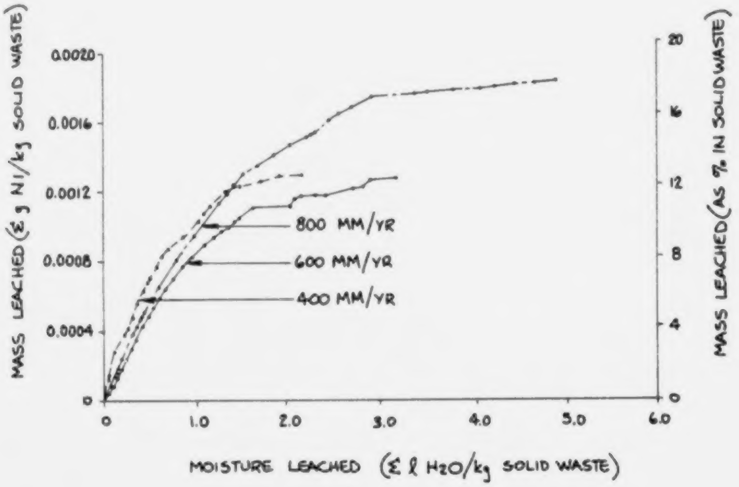

Figure 20. Mass leached: Nickel

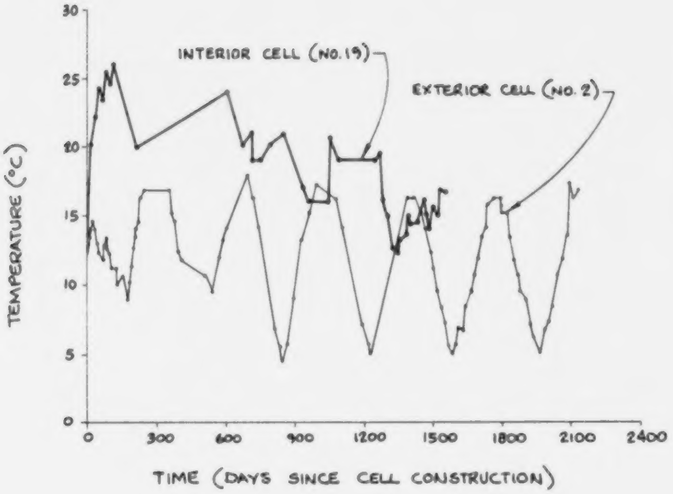

Figure 21. Refuse temperatures.

about 27°C within 50 to 100 days after initial loading. Thereafter, temperatures average about 20°C, although they are subject to large fluctuations. From observation, these fluctuations appear to be somewhat seasonal with highest temperatures occurring in the wintertime. This can be explained by their placement at a higher location within the Center Hill Laboratory's high-bay area and exposure to warmer drafts from the heating system than is experienced in the summer.

The exterior test cell also appears to have climbed in temperature shortly after placement but the peak was reached within 25 to 50 days at only 15°C. Thereafter, temperatures appear to drop to about 8°C and may be partially explained by its location below the ground surface and its construction date of November 1974. Because of its relatively small mass, refuse temperatures are readily affected by soil temperatures. Also, since it was constructed in November 1974, it was shortly thereafter exposed to the cold wintertime temperatures. Thereafter, temperatures appear to range between 4 and 18°C. Again, a seasonal fluctuation is noted; unlike the interior test cells, however, temperature lows occur in the winter, temperature highs in the summer.

GAS EVALUATION

Gas samples are extracted from all test cells on a monthly basis for compositional analysis. In addition, gas quantity data is recorded for interior Test Cells 16, 18, and 19 on a daily basis. Because both quality and quantity data is available for only the interior cells, our attention here will be redirected to these three cells.

Plots of gas composition vs time for Cells 16, 18, and 19 have been included as Figures 22, 23, and 24, respectively. As shown, compositional data is available for each of these cells for periods ranging from 5 1/4 years (for Test Cells 18 and 19) to 5 3/4 years (for Test Cell 16).

Carbon dioxide is the predominant gas created in the early phases of solid waste decomposition. Theoretically, carbon dioxide is generated as the predominant gas during both aerobic decomposition and the facultative (or acid-forming) phase of anaerobic decomposition. Later, as the methanogenic phase of anaerobic decomposition is established, methane becomes the predominant gas although carbon dioxide concentrations should continue to comprise from one-third to one-half of the gas volume.

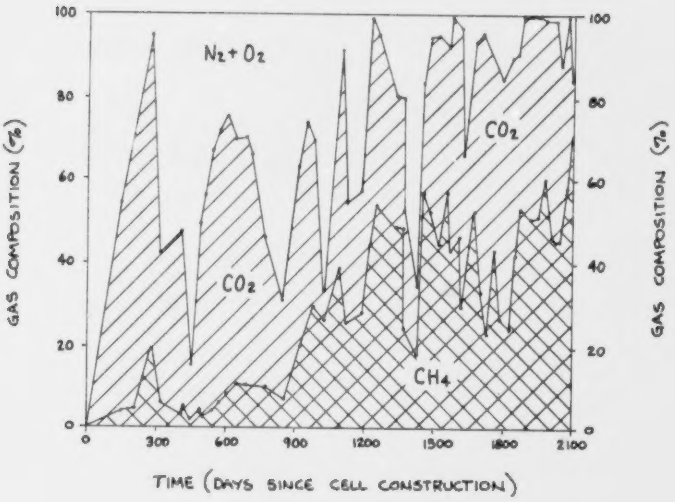

Figure 22. Gas composition at Test Cell 16.

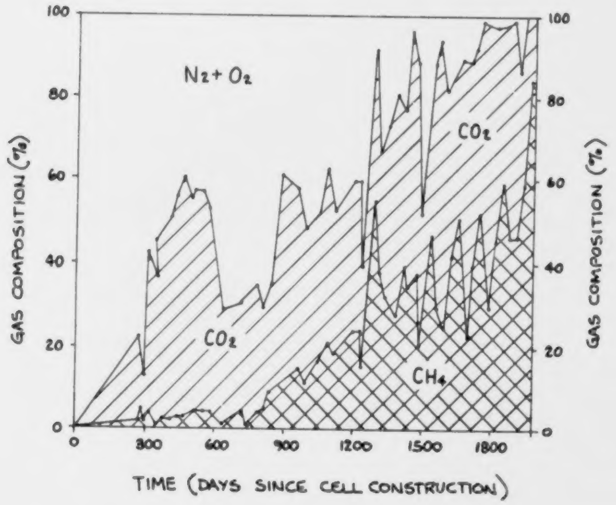

Figure 23. Gas composition at Test Cell 18.

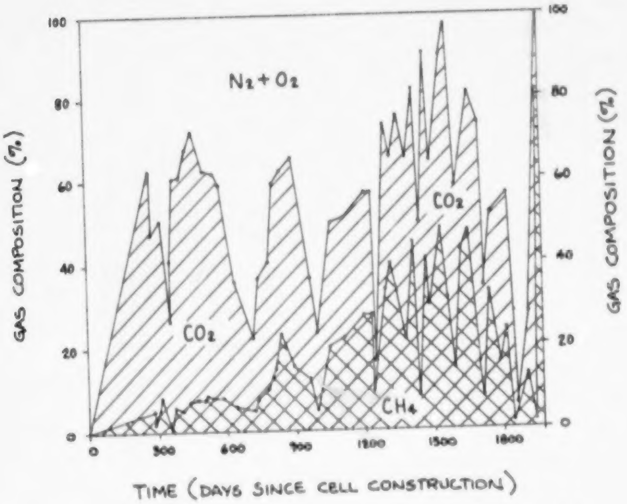

Figure 24. Gas composition at Test Cell 19.

The plots contained in Figures 22, 23, and 24 generally support these theories. Highest carbon dioxide readings were recorded quite early in the decomposition process with readings attaining between 60 and 90 percent for all test cells within 1 to 1 1/2 years after construction. Thereafter, carbon dioxide concentrations appear to decrease as methane concentrations increase. However, carbon dioxide concentrations of 40 to 60 percent are generally still maintained in all three test cells, even five years after cell construction.

As was anticipated, methane concentrations were initially low, averaging no more than five percent over the first two to three years of the study period. After about 2 1/2 years, however, methane concentrations were seen to steadily increase in all test cells and have now attained readings between 50 and 60 percent in all three cells. Typical atmospheric gases (nitrogen and oxygen) consume the balance of gas compositions and were found to contribute up to 80 percent of the gas through the first year before declining steadily to levels between 0 and 20 percent. Other gases (e.g., hydrogen) were also detected over the time period shown, but were in trace concentrations of less than one percent and, therefore, not shown. Overall, it appears that the trends noted in this study for gas composition have generally followed that specified by decomposition theory. The trends shown in Figures 22 through 24 appear to generally follow the theoretical gas production patterns reported by Rovers and Farquhar.[5]

Plots of gas quantity with time (ml gas/kg of solid waste/day) for Cells 16, 18, and 19, have been included as Figures 25, 26, and 27, respectively. First, it should be noted that the gas volumes shown are not necessarily for a precise 24-hour period and, in some cases may represent anywhere from 12 to 48 hours. Nevertheless, these plots provide good data on the overall range and trending of gas generation to date. As shown, gas volumes for all three cells appear to range from 0.5 to 2.0 ml/kg/day for the first 3 1/2 years. Thereafter, daily gas volumes have increased substantially, ranging between 1 and 50 ml/kg/day. Although the ranges of gas generation are similar for all three cells, average gas generation rates are not. Cells 16 and 18 have had average generation rates of about 25 ml/kg/day. Average rates in Cell 19 are considerably lower at about 7 ml/kg/day. The dramatic increase in daily gas volume for all three cells at the 1,200 day point is somewhat unexplained. It should be pointed out that a transition in sampling personnel was made at that time and this transition may be wholly accountable for the increased volumes recorded. Nonetheless, we have no evidence to suggest that a difference in sampling methodology is involved.

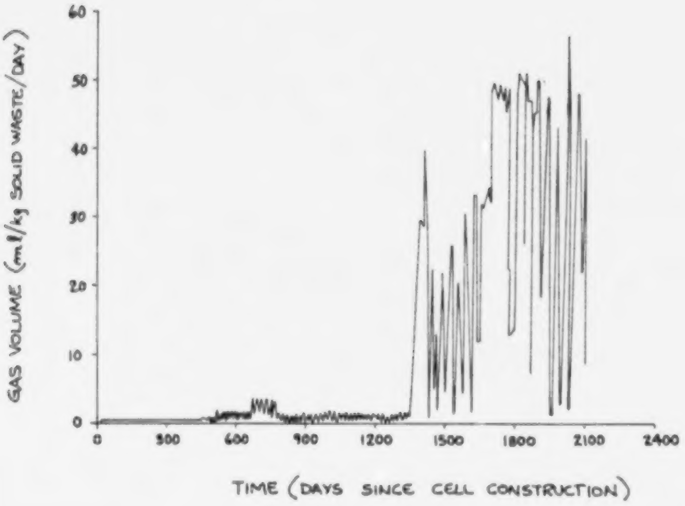

Figure 25. Daily gas volumes at Test Cell 16.

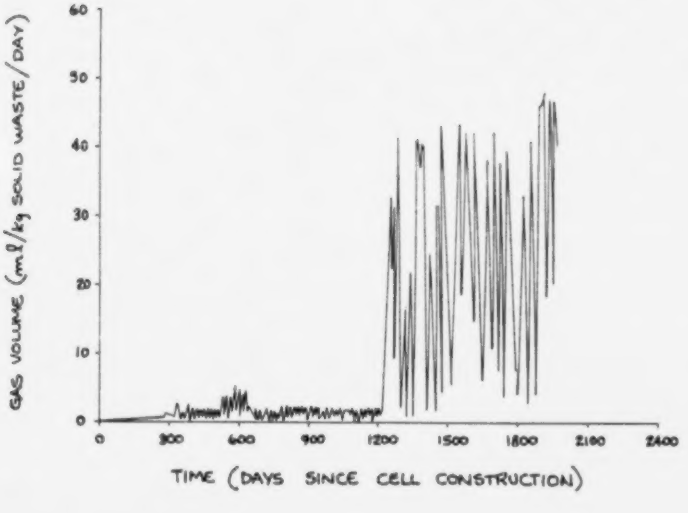

Figure 26. Daily gas volumes at Test Cell 18.

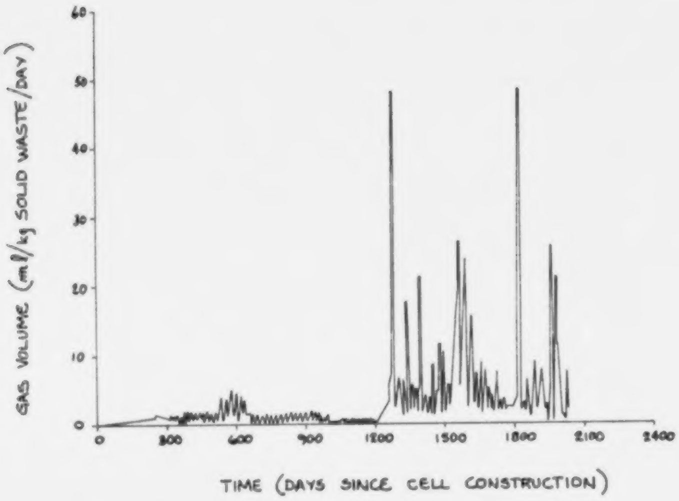

Figure 27. Daily gas volumes at Test Cell 19.

Plots of cumulative gas generation to date for each of these three test cells are contained in Figures 28, 29, and 30. Overall trendings for cumulative gas generated in Cells 16, and 18 are almost identical. Total gas volume achieved is about 18 l/kg of solid waste over 5 3/4 years for Test Cell 16, and about 14 l/kg over 5 1/4 years for Test Cell 18. The overall trending and total cumulative gas volume encountered in Test Cell 19 is considerably less than that for the other cells. This, of course, reflects the lower daily average rates seen in Figure 27. Total cumulative gas volume generated by this Cell was calculated at about 3 l/kg in the 5 1/4 years since its construction.

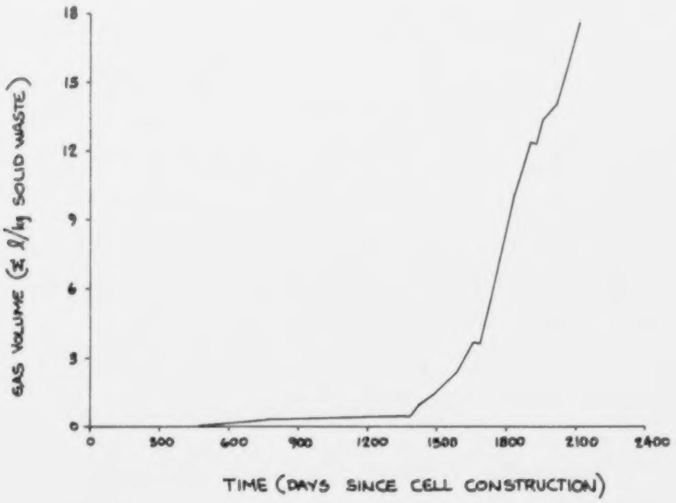

Figure 28. Total gas generation at Test Cell 16.

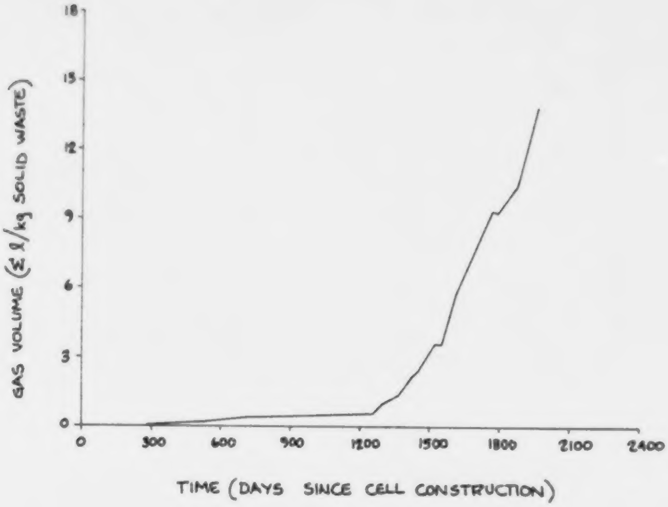

Figure 29. Total gas generation at Test Cell 18.

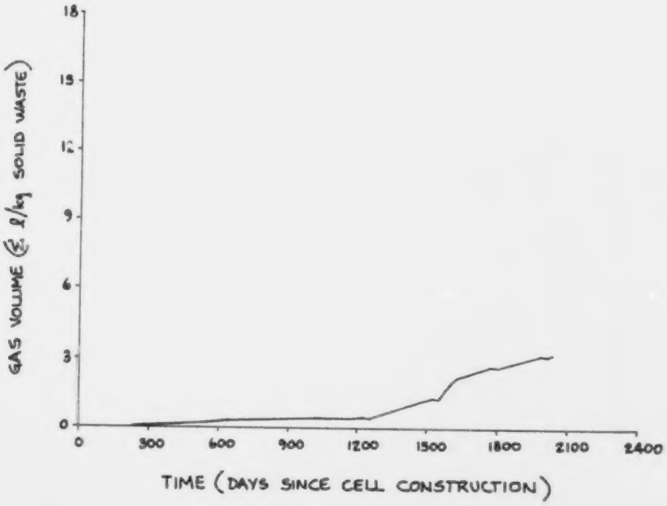

Figure 30. Total gas generation at Test Cell 19.

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Gas Production in Municipal Waste Test Cells

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ABSTRACT

Data on landfill gas volumes and gas composition have been collected from four municipal solid waste test cells for 636 days. The average gas production for the four cells for the 636-day period was 18.32 ml/kg (dry wt) of refuse/day. Gas production averaged over 636 days was only 30% of the rate recorded for the first 100 days. First hydrogen and then methane was detected in the decomposition gases from each cell. Methane concentrations as high as 5.8% have been recorded. The larger test cells used in this study are producing 10% more gas per kg refuse than the smaller cells. The higher gas production is thought to be due to a higher operating temperature in the larger cells.

INTRODUCTION

Landfills can be considered uncontrolled, unmixed, anerobic digestors. Work that has been done on landfill gas generation both in the laboratory and in the field indicates that many factors such as refuse composition, moisture content, degree of compaction, pH, temperature, nutrient, and metal content can affect the rate of decomposition and gas production in a landfill (1,6). Landfill gas production is also controlled by changes in the microbial flora that occur as oxygen is depleted in the decomposing refuse. All of the factors listed above can be thought of as affecting or controlling the microbial activity that occurs in the test cells. The sequence of bacterial communities may be the most critical factor related to gas production. Most organisms involved in anaerobic decomposition with gas generation operate in an interdependent fashion with one organism using the metabolic products of another for essential nutrients (8).

The present study was developed to verify gas production data obtained in other refuse decomposition studies and to examine problems related to scaling of large laboratory simulation tanks. Data are presented on gas composition, and gas volumes for the first 636 days of operation. The data are compared with results from other published studies.

METHODS AND MATERIAL

The experimental setup and analytical procedures used in this investigation are described in detail in Myers and others (7). Two different sized gas-tight steel test cells were used. The larger tanks have an inside diameter of 1.83 m and a height of 3.66 m. The smaller tanks have an inside diameter of 0.91 m and a height of 1.83 m. The tanks were packed with unbailed, unshredded municipal waste from residential collection routes in Warren County, Mississippi. The composition of the refuse is given in Figure 1. Details of the loading and operation of the test cells are given in Table 1. Cross-sections of tanks are shown in Figure 2. Infiltration of precipitation was simulated by adding 1.27 cm of deionized water to each tank per week. This mimics the infiltration that could occur in a landfill in the humid eastern U.S. Leachate is drained from the tanks and sampled at monthly intervals.

An automatic gas measuring system recorded the volume of gas produced and the temperature and barometric pressure at intervals throughout the 636-day experiment (Fig. 3). Details of the measuring and data reduction system are given in Myers

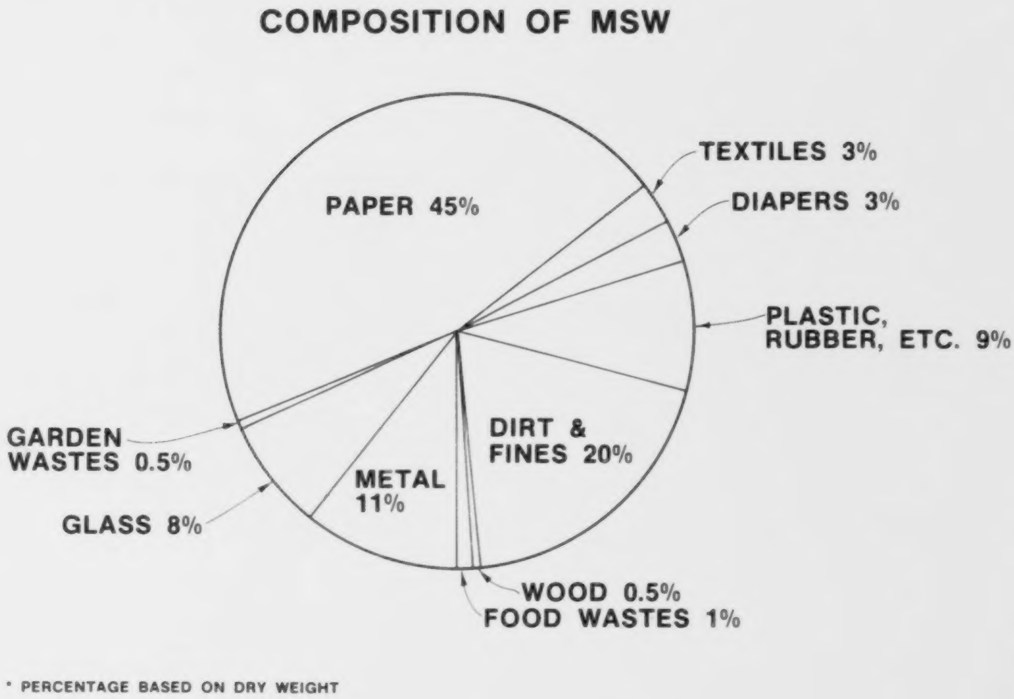

Figure 1. Composition of refuse.

TABLE 1. SUMMARY OF DATA ON TEST CELLS

	Test Cells			
	10	15	16	22
Inside diameter (m)	0.91	1.83	1.83	0.91
Height (m)	1.83	3.66	3.66	1.83
Wet weight of refuse (kg)	322	2555	2555	318
Dry weight of refuse (kg)	264	2096	2096	261
Density of refuse (kg/m^3)	405	400	400	405
Cover Material	gravel	gravel	gravel	gravel
Underlying soil	clayey sand	clayed sand	clayey sand	clayey sand
Surface area of refuse (m^2)	0.65	2.63	2.63	0.65
Weekly moisture addition (cm)	1.27	1.27	1.27	1.27
Volume of water added at (l) weekly interval	8.3	33.4	33.4	8.3

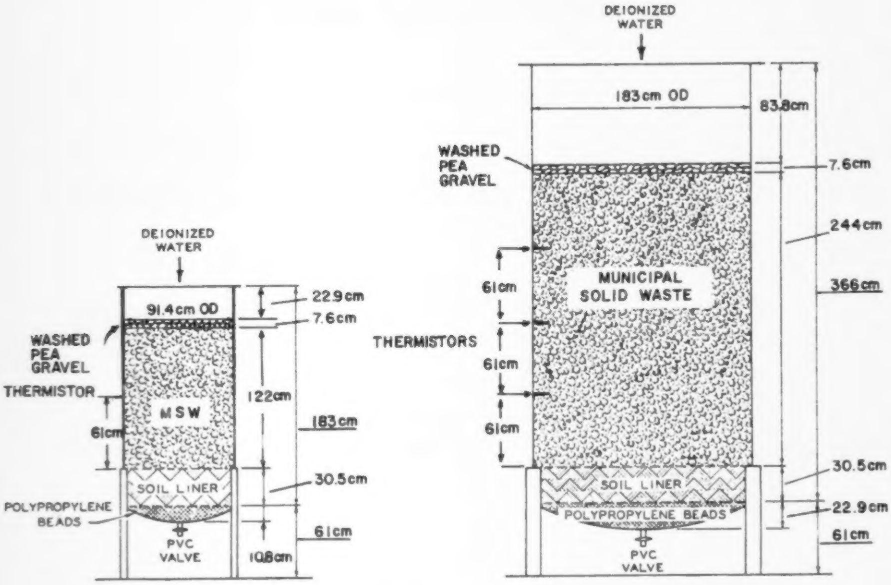
DEIONIZED WATER
91.4cm OD
22.9cm
7.6cm
WASHED PEA GRAVEL
THERMISTOR
MSW
61cm
122cm
183cm
30.5cm
SOIL LINER
POLYPROPYLENE BEADS
PVC VALVE
61cm
108cm
DEIONIZED WATER
183cm OD
83.8cm
WASHED PEA GRAVEL
7.6cm
244cm
MUNICIPAL SOLID WASTE
366cm
61cm
THERMISTORS
61cm
61cm
30.5cm
SOIL LINER
POLYPROPYLENE BEADS
22.9cm
61cm
PVC VALVE

Figure 2. Cross-section of refuse test cells.

and others (7). Gas samples were taken periodically and analyzed on a Perkin-Elmer Sigma 3 gas chromatograph for oxygen, hydrogen, nitrogen, carbon dioxide and water vapor. Gas analyses discussed here include only those for hydrogen and methane.

RESULTS AND DISCUSSION

Quantity of Gas Produced

The total gas production for landfill simulators is given in Table 2. The average gas production for all four test cells was 18.32 ml/kg (dry wt)/day. This is only 30 percent of the average rate recorded for the first 100 days.

These gas production figures are higher than values reported in other studies involving test cells maintained for comparable lengths of time. Chian and DeWalle (2) reported a maximum rate of gas production of 15 ml/kg (dry wt)/day for a 400-day experiment using smaller (208 l) test cells. In data running from 912 to 1280 days into a simulation experiment Walsh and Kinman reported that the highest gas production figure was only 1.0 ml/kg (dry wt)/day (9). Unfortunately no data are available for the first 600 days of the experiment due to difficulties encountered in sealing the cells. Most test cells will underestimate gas production due to leakage or equipment failure. Actual production of gas in a fill has been estimated at 22-45 ml/kg/day (3). The higher gas production rates in this study may reflect the increased accuracy produced by the automated gas measuring system used in the present equipment.

Gas Composition

In an earlier report covering the first 100 days of test cell operation it was noted that; oxygen levels in all four test cells declined rapidly and that the percentage of CO_2 increased in all four tanks. Nitrogen levels declined gradually over the first 100 days due to sweeping of the tanks by CO_2 produced in decomposition. No hydrogen or methane was noted in this early part of the study.

Both methane and hydrogen appeared in all of the test cells (except no. 16) by

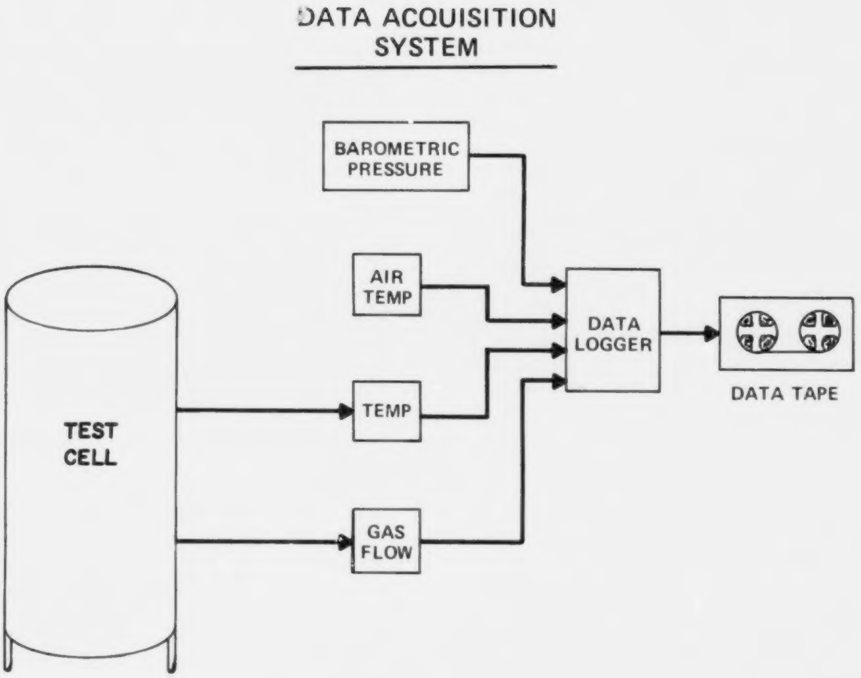
DATA ACQUISITION SYSTEM
BAROMETRIC PRESSURE
AIR TEMP
TEST CELL
TEMP
GAS FLOW
DATA LOGGER
DATA TAPE

Figure 3. Schematic of gas data acquisition system.

TABLE 2. GAS PRODUCTION FROM TEST CELLS FOR 636 DAYS OPERATION

Test Cells No.	Total Gas (1)	Methane (1)	Hydrogen (1)
10	2928.2	15.1	4.8
15	25937.2	86.3	196.0
16	26018.2	16.4	223.4
22	2813.2	17.1	3.8

the 250th day of operation. The daily average production (taken over ten-day intervals) of hydrogen and methane for each of the tanks are given in Figures 4-7. All of the test cells showed the typical gas production pattern proposed by Farquhar and Rovers (5) with appearance of hydrogen preceding the development of methane. In all cases hydrogen production has dropped off as methane production increased. This is due to the consumption of hydrogen in reduction of carbon dioxide to produce methane (6). In the test cell 16 where hydrogen production was highest methane production was lowest, suggesting inhibition of carbon dioxide reduction (Table 3) was occurring.

Gas analysis were routinely made at 30 day intervals, peak methane and hydrogen levels reported here are spot samples of gas from the head space of the tank on the sampling date. The highest methane concentration was noted as 8.9% in tank 22 after 625 days. The highest hydrogen concentrations (5.8%) appeared in tank 16 after 282 days. The maximum methane and hydrogen concentrations compare well with data obtained by Engineering-Science (4). in instrumentation on an actual landfill. Typical analyses of gas from the bottom of the test landfill (Probe 33 A) showed 7.3% methane after 648 days of decomposition. Hydrogen concentration at this same probe was up to 5.5% after 358 days decomposition. While a great deal of variation exists in landfill gas composition data the results from the test cells are reasonable when compared to the composition of actual landfill gas.

Scaling of Experiment - Two sizes of test cells were incorporated into the gas generation program so that the effects of any possible scaling problem could be noted. The different tank designs allow the smaller tanks to saturate more quickly due to the shorter column of refuse involved and the smaller tank should equilibrate more rapidly with surrounding temperatures. At this stage in the experiment, both of the larger tanks show a slightly higher average gas production rate (∿10% larger) than the smaller tanks. Thermistor data indicate that the larger test cells are at a slightly higher temperature than the small test cells. Temperature differences may be due to the slower heat dissipation from the larger cells or the thermal stratification in the building. The higher temperatures could account for more rapid decomposition and greater gas production.

SUMMARY AND CONCLUSIONS

Data on gas composition and gas volumes generated from four landfill test cells have been collected for 636 days. The average gas production for the 636-day period was 18.32 ml/kg (dry wt)/day. This is only 30% of the average production rate observed during the first 100 days of operation. Methane and hydrogen have both appeared in the gas emitted from the test cells. Hydrogen appeared in each tank before methane was detected. The highest methane concentration observed to date was 8.9%. The highest hydrogen concentration was 5.8%.

TABLE 3. GAS PRODUCTION FOR TEST CELLS CORRECTED FOR WEIGHT OF REFUSE

Test Cell No.	Volume of Total Gas Produced per kg (dry wt) of refuse (1)	Volume of Methane Produced per kg (dry wt) of refuse (1)	Volume of Hydrogen Produced per kg (dry wt) of refuse (1)
10	11.09	0.06	0.02
15	12.34	0.04	0.09
16	12.41	0.01	0.11
22	10.78	0.06	0.01

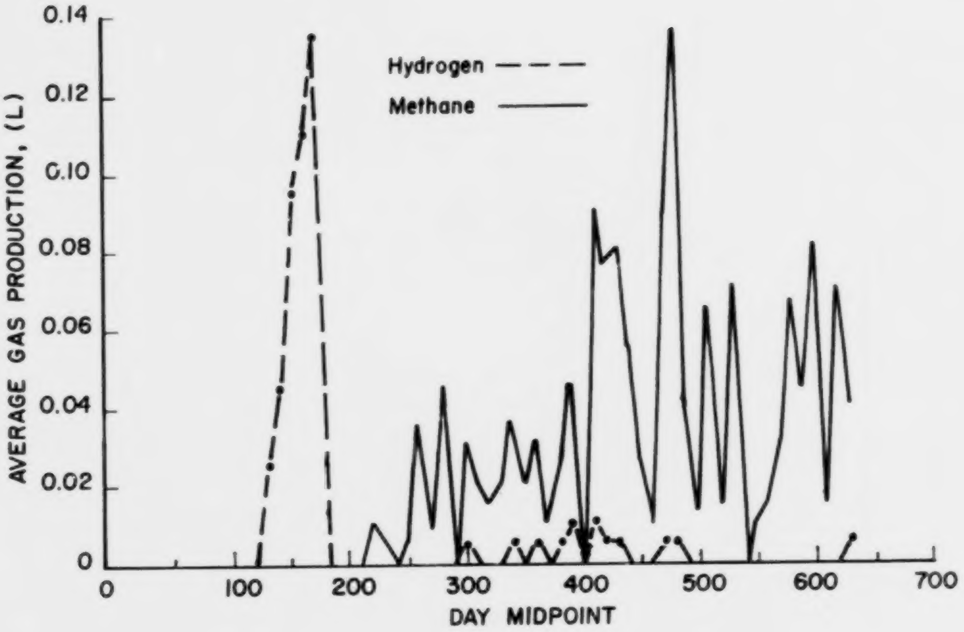

Figure 4. Daily average hydrogen and methane production (over 10-day intervals) for Test Cell 10.

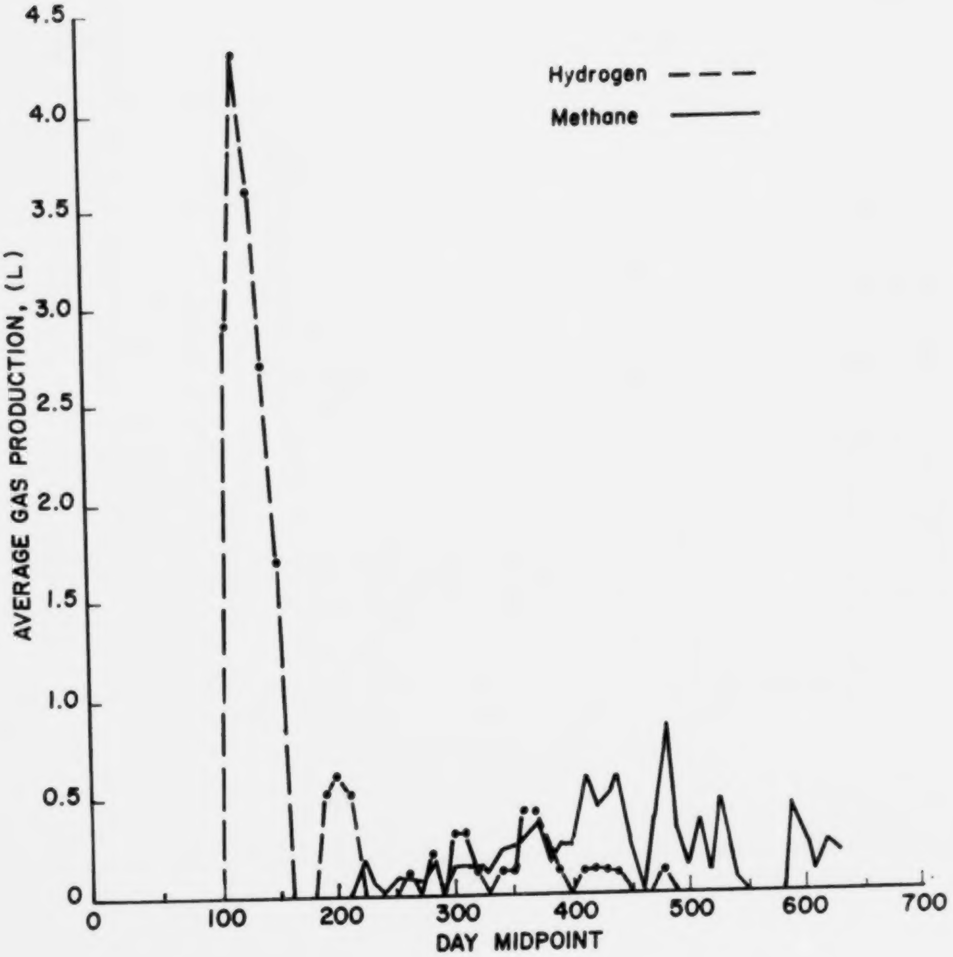

Figure 5. Daily average hydrogen and methane production (over 10-day intervals) for Test Cell 15.

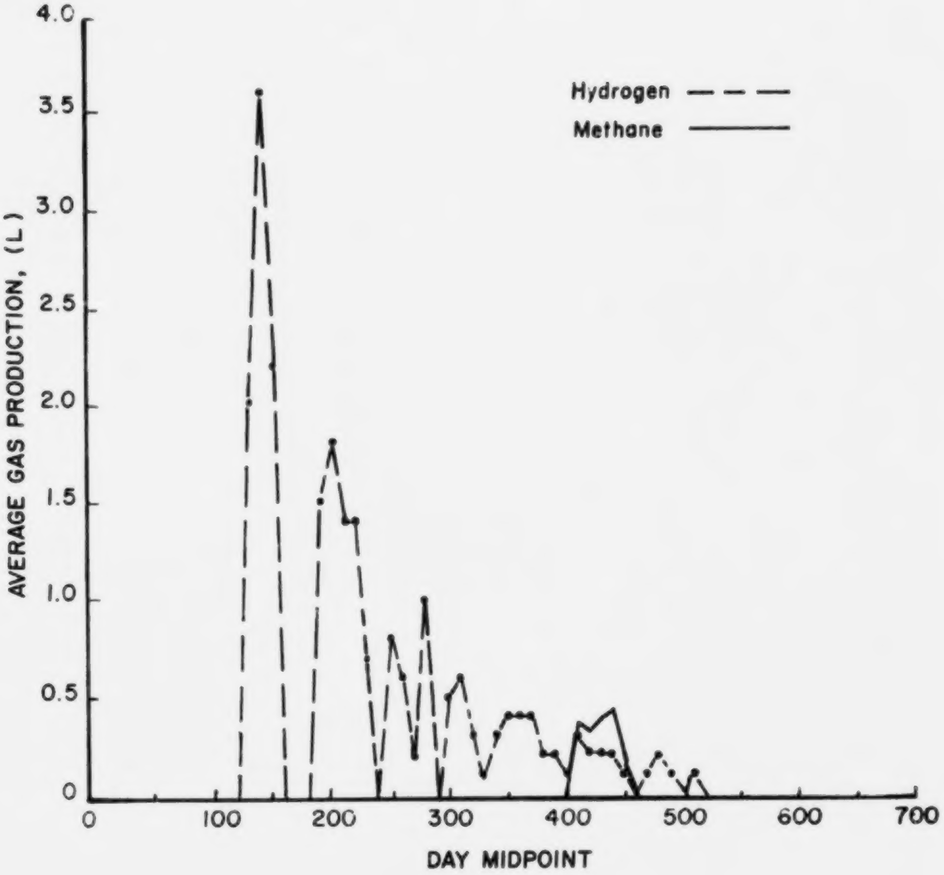

Figure 6. Daily average hydrogen and methane production (over 10-day intervals) for Test Cell 16.

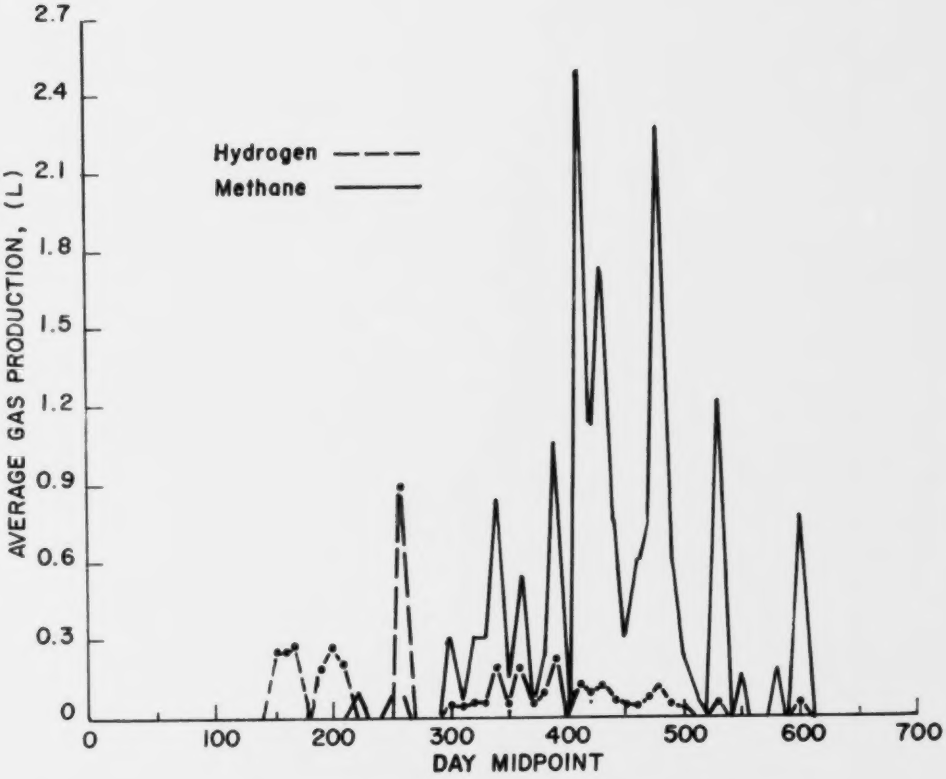

Figure 7. Daily average hydrogen and methane production (over 10-day intervals) for Test Cell 22.

The larger test cells are producing 10% more gas than the smaller cells (on a ml/kg of dry refuse/day) basis. This effect is thought to be due to the increased microbial activity in the larger tank due to slightly higher temperatures.

The tanks are continuing to operate in a satisfactory manner. The gas yields measured are in agreement with data from similar experiments in test cells and landfill field investigations.

ACKNOWLEDGEMENTS

This report is part of a continuing research program on solid and hazardous waste disposal, which is now being conducted by the U. S. Army Engineer Waterways Experiment Station and funded by the U. S. Environmental Protection Agency, Municipal Environmental Research Laboratory, Solid and Hazardous Waste Research Division, Cincinnati, Ohio, under Interagency Agreement EPA-IAG-D4-0569. Robert E. Landreth is the EPA program manager for this work.

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FIELD VERIFICATION OF LANDFILL METHANE MOVEMENT AND METHANE CONTROL SYSTEMS

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ABSTRACT

This paper describes field studies being conducted to verify computer models developed to predict methane transport around sanitary landfills and to provide design support for methane migration control systems.

Introduction

This paper presents a summary of the work completed through December 1, 1980 on the Field Verification of Methane Movement Functions and Methane Control Systems for Landfills study awarded by the Negotiated Contracts Branch of the U.S. Environmental Protection Agency under Contract No. 68-03-2849. The contract/award is being monitored by the Ohio State University Research Foundation (OSURF) under the direction of Dr. Charles A. Moore who serves as principal investigator. The OSURF designation for the project is RF 712366 U.S. EPA. This report is organized to discuss the progress of the site selection, field verification, and analytical (computer) simulation portions of the study and includes a discussion of work currently planned for completion of the study.

Site Selection

The Field Verification of Methane Movement Functions and Methane Control Systems for landfills proposal specifies that three landfill sites be selected for field verification of the computer models developed at Ohio State University (OSU) for gas flow within soil. An additional site is to be selected for field verification of the relative effects of methane migration control systems as predicted by the computer model. This site may or may not be one of the three sites mentioned above. At the time of this report, two sites have been selected for field verification of the computer model. The criterion for selection of these sites are discussed in the following paragraphs.

Criteria for Selection of Landfill Sites

The landfill sites selected for study were evaluated according to the following criteria:

1. surface of the soil surrounding landfill should not be sealed,
2. cohesionless soil should surround the landfill,
3. homogeneous soil should surround the landfill,
4. the groundwater table should be below the landfill,
5. at least three acres should adjoin the landfill,
6. no burning of waste should have occurred,
7. access for constructing gas control facilities should exist, and
8. power should be available.

In all, 40 landfill sites were considered, from which two sites have been selected. There were several additional sites that rated very high, and the final selection from among the several sites was based upon achieving geographical distribution. The following sections present detailed information for each of the chosen sites.

Lees Lane Landfill, Louisville, Kentucky

This site was presented by the firm of Stearns, Conrad and Schmidt (SCS) Engineers. The Lees Lane Landfill is located on a 125-acre tract in Jefferson County, Kentucky as shown in Figure 1. The site is located along the Ohio River and has 4600 feet of river frontage. The balance of the landfill perimeter includes 800 feet of property line abutting Borden Inc. (a chemical manufacturer); 3500 feet of property line abutting Riverside Gardens (a residential development); and 3500 feet of property line abutting Louisville Gas and Electric (a power plant). As shown in Figure 1, a floodwall right-of-way fringes the property line abutting Riverside Gardens. The site was first used as a soil borrow pit in the early 1950's. Excavations to depths of 40 feet proceeded from the northern to the southern end of the site over a period of many years.

Waste was first received at the site in 1952. Waste was deposited in excavated areas and deposition generally proceeded along with the excavation operation (north to south). From 1952 until 1955 the site served as the primary waste disposal site for the City of Louisville and Jefferson County. In 1955 an incinerator was opened by the City of Louisville, and all waste generated by the City was hauled via municipal haulers to the incinerator site. The Lees Lane Landfill continued as the primary waste disposal site for Jefferson County until 1962. During this period, private haulers serving residences and industries in the County hauled collected waste to the site. Much of the industrial waste was from an area located five miles northwest of the landfill where the rubber industry predominated.

In 1975, receipt of wastes at the Lees Lane Landfill was discontinued. Filling proceeded to original grade in the northern two-thirds of the property. Although excavations to depths of 40 feet had proceeded to the southern property lines, very little waste had been deposited in the southern one-third of the property and this remains an excavated pit to this day. All waste deposited at the site was ultimately covered with native soil material.

Riverside Gardens is a residential development located east of Lees Lane Landfill (Figure 1). In March 1975, several flash fires were reported around homes in this development. The City-County Health Department monitored several homes and methane levels up to 20 percent were detected.

The Health Department drilled a series of four monitoring wells in Riverside Gardens in an east-west line perpendiclar to the landfill boundary. These wells are known as W-1, W-2, W-3, and W-4, and are located as shown in Figure 1. The percent and pressure of methane gas in these wells were measured in March and April, 1975. Analysis of the data indicated that the gas had moved through the subsurface an average of 765 feet and a maximum of 860 feet from the landfill boundary.

In April 1975 regular monitoring of these gas wells ceased. However, an inspection was made of these wells in December 1977 by SCS Engineers. At that time, the valve of Well W-2 was opened and an audible pressure release was noted. Since one (and perhaps all) of these wells continued to be under pressure, generation of landfill gases is presumed to be continuing. Further, since these gases are migrating large distances, it is likely that the gas generation rate is relatively high.

In June, 1978 SCS Engineers was awarded a contract by the Jefferson County Department of Public Works to monitor the presence of landfill gases in subsurface areas immediately surrounding the Lees Lane Landfill.

At the onset of the project 14 monitoring wells were drilled. All wells were outside of the fill area, approximately 300 feet from the edge of the refuse. The boring logs for these wells reveal that the soil is a relatively homogeneous mixture of medium sand with some gravel and smaller quantities of silt and clay. Groundwater is generally at a depth of 50 feet below the ground surface.

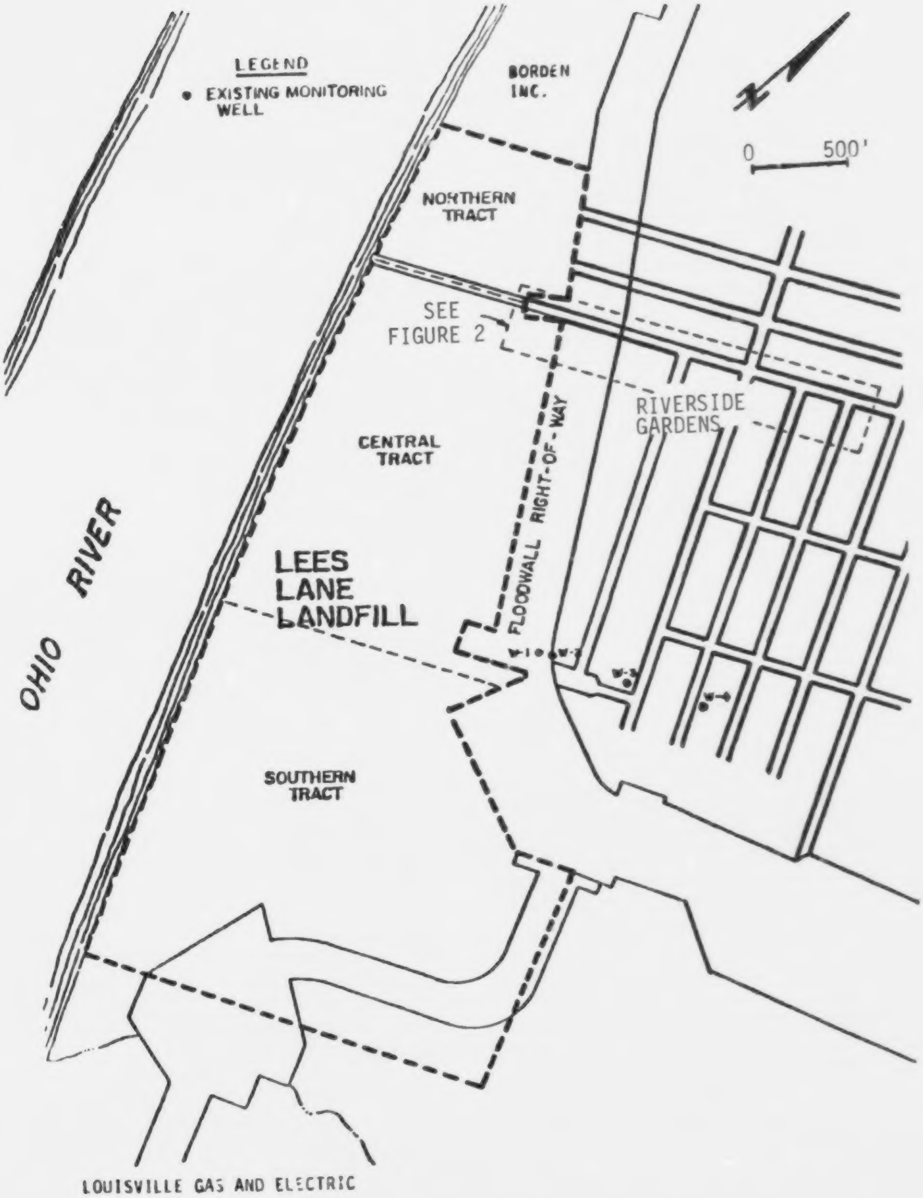
LEGEND
● EXISTING MONITORING WELL
BORDEN INC.
0 500'
NORTHERN TRACT
SEE FIGURE 2
RIVERSIDE GARDENS
CENTRAL TRACT
FLOODWALL RIGHT-OF-WAY
OHIO RIVER
LEES LANE LANDFILL
W-1
W-2
W-3
W-4
SOUTHERN TRACT
LOUISVILLE GAS AND ELECTRIC

Figure 1. Site Map of Lee's Lane Landfill, Louisville, Kentucky

The results of the gas monitoring program completed by SCS at Lees Lane Landfill prompted the Jefferson County Health Department to install a gas migration control system along the eastern portion of the site. Subsequently, the county contracted with SCS to design and install the system. The system has been operating at Lees Lane since October 1980.

The Lees Lane Landfill exhibits good conformance with the site selection criteria listed above.

1. The ground surface over the landfill is not sealed. There is no standing water on the site and all refuse fill areas are covered only by low-lying vegetation and grass. Most of the area surrounding the landfill is grass-covered or wooded.
2. The soil is predominantly of medium sand with some gravel and lesser amounts of clay and silt . Therefore, the soil is relatively cohesionless.
3. The soil is relatively homogeneous to a depth of 50 feet. (The landfill extends to a depth of 40 feet.) Textural changes consist of spatial variations in clay and silt content. Granular soils predominate throughout the site.
4. Groundwater is approximately 10 to 15 feet below the bottom of the landfill. The ground water is not in a confined aquifer, and there are no perched water tables.
5. The land located to the west of the landfill is accessible and has an area greater than three acres. It is possible to locate an area of the landfill which was filled within two years time and which was closed from eight to 25 years ago.
6. According to County officials, burning was not an operational regularity at the site and when fires did occur, they were quickly extinguished.
7. A gas control system has been installed at the site. This system may be turned on or off during the present study to allow the examination of gas migration from an initial "day zero" simulation.
8. Power is available at the site.

Mississauga Landfill, City of Mississauga, Regional Municipality of Peel, Ontario, Canada

This site was presented by Morrison Beatty Limited, Etobicoke, Ontario. The site is located within the regional municipality of Peel in the southeast limits of the City of Mississauga at latitude 43° 31' N. and longitude 79° 37' W. The landfill is approxmately 100 acres in size and is bordered by Springbank Road to the north, Mississauga Road to the west, North Sheridan Way to the south and Queen Elizabeth Way to the east as shown on Figure 2. Private residences adjoin the site to the east while apartments are contiguous with the site on the west.

The landfill was operated by Berrill and Trustrum Limited, a private contractor. It served the City of Mississauga which has a population of over 200,000. The refuse fill consists of approximately 80 percent domestic refuse and garbage, and 20 percent commercial wastes and industrial refuse. No liquids, sludges or hazardous wastes have been disposed at the site and the burning of refuse was not practiced.

The landfill is located in two worked out sand and gravel pits. The smaller of the two sites is 10 acres in area and about 35 feet deep. Its capacity is on the order of 150,000 tons, and it was filled between April 1971 and September 1972. The larger site is approximately 90 acres in area and extends from 30 to 50 feet below the ground surface. Filling began in 1966 and was completed in March 1980. The first area to be filled was the northeast corner of the site near Springbank Road South and Queen Elizabeth Way. Filling proceeded in a general south west direction. The area near Springbank Road North was filled between 3 and 4 years ago. It is in this area that the field verification study is being conducted.

The Mississauga landfill site is developed within the Iroquois Plain physiographic region. This region is characterized by a gently sloping sand plain that averages about 2 miles in width, paralleling the present Lake Ontario shoreline. The sand deposit marks the shoreline of post-glacial Lake Iroquois.

The sand and gravel deposits at the site were laid down in the form of a barrier beach where a river emptied into Lake Iroquois. The deposits are indicative of those laid down in moderate to fast flowing waters. There is very little silt or

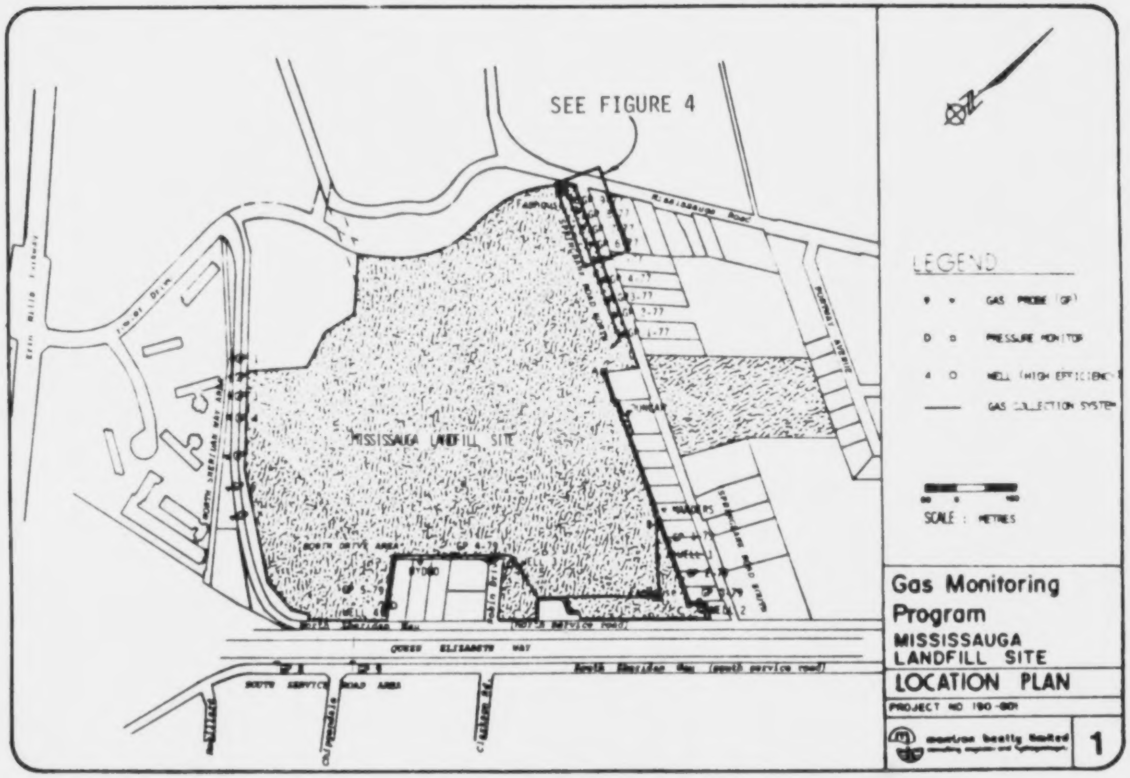
SEE FIGURE 4
MISSISSAUGA LANDFILL SITE
Mississauga Road
Queen Elizabeth Way
South Sheridan Way (south service road)
North Service Road
South Service Road Area
North Sheridan Way Area
North Drive Area
Erin Mills Parkway
Springbank Road North
Springbank Road South
Burnway Avenue
GP 4-77
GP 3-77
GP 2-77
GP 1-77
GP 5-79
GP 4-79
WELL 6
WELL 2
LEGEND
GAS PROBE (GP)
PRESSURE MONITOR
WELL (HIGH EFFICIENCY)
GAS COLLECTION SYSTEM
SCALE : METRES
Gas Monitoring Program
MISSISSAUGA LANDFILL SITE
LOCATION PLAN
PROJECT NO 190-801
morrison beatty limited
1

Figure 2. Site Map of Mississauga Landfill, Ontario, Canada

clay in the material. The deposit is predominantly medium to coarse sand with lenticular deposits of gravel. There is a zone of cemented gravel which is generally encountered in the upper 9 feet.

Off-site methane migration was first monitored in 1971 when explosive concentrations were detected in the basements of several houses near the fill. Severe vegetation stress also occurred in yards contiguous with the landfill. The preliminary monitoring showed the 50 percent methane contour was more than 80 feet from the landfill boundary. Detectable methane concentrations were monitored in gas probes 200 feet from the landfill.

Under the direction of the Regional Municipality of Peel, a gas interceptor system was installed in 1972 along Springbank Road South and Queen Elizabeth Way to protect the affected properties. Initially, this system was run on an intermittent basis. However, in 1977 a burner unit designed to flare off the gases exhausted from the system was installed. The interceptor system has run continuously since that time.

This system was extended in stages as the landfill operation expanded. Gas probes were also installed along the interceptor system to monitor its effectiveness. The locations of these probes are shown on Figure 3. The collection system along Springbank Road North was completed in September of 1977. This system has run continuously since February of 1979.

Morrison and Beatty has monitored the site for the Region of Peel since explosive methane concentrations were first detected in 1971. Static pressure readings and percent combustible gas data were recorded. The frequency of this monitoring varied over the years, but typically data was taken 4 or 5 times during the winter months and twice per month during the summer. Results of gas chromatographic analyses are also available for some of the probes.

Officials with the Regional Municipality of Peel have planned an additional gas interceptor system which will be installed in the spring of 1981. This system will run along the western and southern edges of the fill. Any probes that will be installed to gauge its effectiveness may also be of use to this verification project.

The Mississauga Landfill Site exhibits good conformance with the site selection criteria listed previously.

1. The ground surface of the site is not sealed. However, the long winter season usual in the landfill area may freeze the upper soil layer and effectively seal the ground surface.
2. The soil is predominantly sand with some silt, clay and gravel mixed in. Therefore, the soil is relatively cohesionless.
3. The soil is relatively homogeneous to a depth of 55 feet. (The Landfill extends to a depth of 47 feet). Textural changes consist of spatial variations in clay and silt content. Granular soils predominate throughout the site.
4. Groundwater is approximately 5 to 10 feet below the bottom of the landfill. The ground water is not in a confined aquifer. No perched water tables exist at the site.
5. The land located to the north (see inset, Figure 2) and west of the site is accessible and has an area greater 3 acres.
6. According to our information, burning was not an operational regularity at the site.
7. A gas control system has been installed at the site. This system may be turned on or off during the present study to allow the examination of gas migration from an initial "day zero" simulation. The Canadian Ministry of the Environment and the Region of Peel have consented to cooperate fully with the "day zero" simulation.
8. Power is available at the site.

Selection of Additional Site(s)

The selection of the third site for inclusion in the field verification portion of this project is currently proceeding. We are investigating seven sites in the Denver, Colorado Metropolitan Area for inclusion in this study. The Department of Energy is currently funding methane research in the Denver area, specifically within Adams County and Commerce City, Colorado, and it is hoped that the inter-

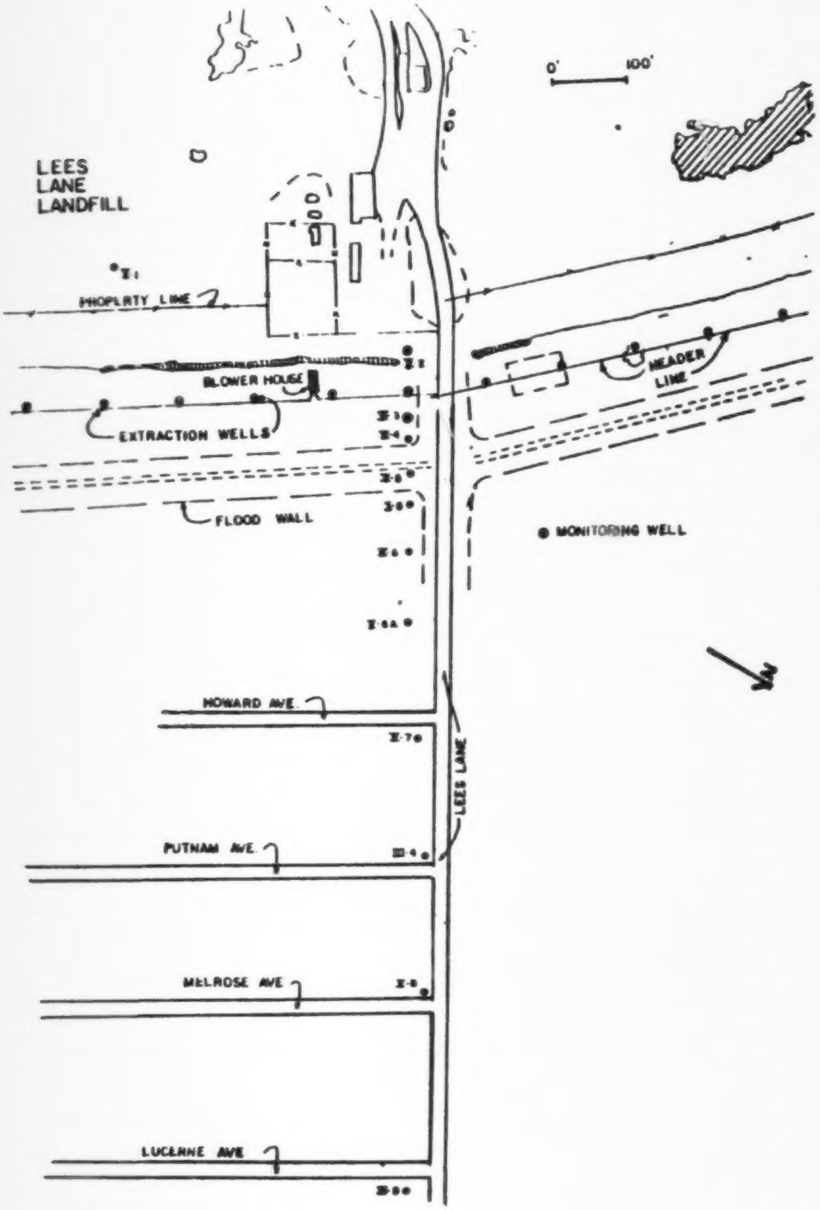
LEES
LANE
LANDFILL
0'
100'
PROPERTY LINE
BLOWER HOUSE
EXTRACTION WELLS
HEADER LINE
FLOOD WALL
● MONITORING WELL
II-1
II-2
II-3
II-4
II-5
II-6
II-6A
HOWARD AVE.
II-7
LEES LANE
PUTNAM AVE.
MELROSE AVE
LUCERNE AVE
N

Figure 3. Location of Monitoring Wells at Lee's Lane Landfill

est in methane research in Colorado would contribute to the progress of this study. We anticipate that the third site for study will be selected by January 1, 1981.

As mentioned previously, one site is to be selected for evaluation of methane migration control systems. At this time, we anticipate that one of the three sites used for the field verification of the computer model will also be used for this portion of the study. We discussed the possibility of using the Mississauga site with representatives of the U.S. EPA and the Canadian Ministry of Environment in Cincinnati late in July. However, no decision regarding this matter has yet been made. We are considering the migration control study in our investigation of possible landfill sites in Colorado. We believe that this will provide an alternative for comparison with the Ontario site.

Field Verification of Methane Movement Predictions

The field verification of computer predictions of methane movement is in progress. Gas concentration monitoring wells have been installed at both the Lees Lane and Mississauga sites. SCS and Morrison-Beatty are currently monitoring gas concentrations at each of the respective sites on a periodic basis. As we anticipated in the Technical Proposal, the monitoring wells were installed within the boreholes required for subsurface investigation at each site to realize a cost savings. A discussion of the results of completed work at each site is contained within the following paragraphs.

Lees Lane Landfill, Louisville, Kentucky

Ten nests of monitoring wells were installed in the period June 24 through July 15, 1980 along Lees Lane at the locations indicated on Figure 4. Each nest consisted of gas probes installed at depths of approximately 15, 30 and 50 feet in three adjacent boreholes. Exceptions are nest V-1 which consisted of three probes installed in a single borehole at depths of 5, 10 and 18 feet and nest V-6 which consists of three probes in a single borehole at a depth of 10 feet.

The wells for the probes were drilled to the desired depth with a hollow stem auger. Probes were then installed generally by the following procedure:

1. A 1 1/2-inch PVC tubing was placed down the center of the auger. The tubing was slotted every 3 inches within the 3 foot section at the desired probe depth. The slotted section was wrapped with cloth towels to prevent clogging of the slots. In addition, tubing in deep wells was slotted at the end to allow a measurement of groundwater fluctuations.
2. The auger was pulled completely from the well. Some caving of natural materials occurred in the deep wells.
3. Backfill consisting of the auger spoil was added around the tubing to fill the well up to the bottom of the three foot slotted section.
4. Three feet of pea gravel was placed around the slotted section of the tubing.
5. A three foot bentonite plug layer was placed above the pea gravel.
6. The well was backfilled to within 6 feet of grade with auger spoil and another 3 foot bentonite layer was placed above the auger spoil.
7. The well was brought to grade by placing concrete above the bentonite plug. A cast iron pipe approximately 18 inches long was placed around the PVC tubing and into the concrete to provide protection for the well installation. Both the PVC tubing and cast iron pipe were capped.

In addition to the ten nests installed for this study, two gas monitoring wells installed previously by SCS Engineers will be monitored during the field investigation. These wells are designated on Figure 4 as wells I-6 and III-4.

The subsurface conditions encoutered during the installation of gas monitoring wells at Lees Lane Landfill were somewhat variable.

The soils consisted generally of 7 to 15 feet of clayey silt underlain by granular soils. The granular soils consisted of interlensed medium dense silty sand, medium dense sand, and medium dense sand and gravel. The granular soils extended to the maximum depth explored of 62 feet. Ground water was encountered generally at depths of 50 to 58 feet below the ground

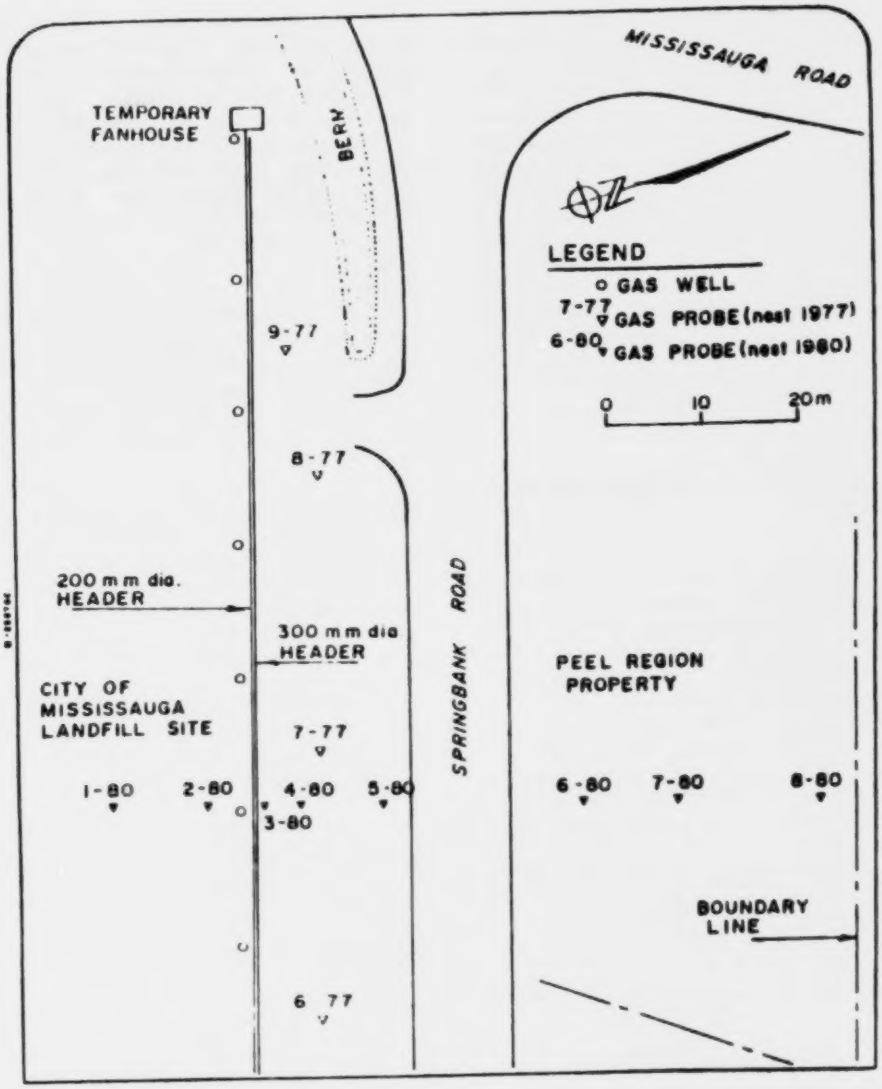
MISSISSAUGA ROAD
TEMPORARY FANHOUSE
BERM
LEGEND
○ GAS WELL
7-77 ▽ GAS PROBE (nest 1977)
6-80 ▼ GAS PROBE (nest 1980)
0
10
20m
9-77
8-77
200 mm dia. HEADER
300 mm dia HEADER
SPRINGBANK ROAD
PEEL REGION PROPERTY
CITY OF MISSISSAUGA LANDFILL SITE
7-77
1-80
2-80
3-80
4-80
5-80
6-80
7-80
8-80
BOUNDARY LINE
6 77

Figure 4. Location of Monitoring Wells at Mississauga Landfill

surface. Test Boring V-1, located within the landfill, encountered 5 feet of clayey silt fill underlain by municipal waste fill to a depth of 18 feet.

Mississauga Landfill, City of Mississauga, Ontario

Eight nests of monitoring wells were installed in the period June 10 through June 13, 1980 at the locations indicated on Figure 5. Each nest consisted of gas probes installed at depths of approximately 15, 30, and 50 feet in three adjacent boreholes. Nests 1-80, 2-80 and 3-80 were located within the existing landfill.

The wells for the probes were drilled to the desired depth with a hollow stem auger. The probes were then installed by the following general procedures:

1. The auger was pulled 6-12 inches and 3/4-inch PVC tubing was placed inside the auger. The tubing was slotted every 3 inches a distance of 3 feet from the bottom of the tubing in shallow (15 foot) and medium (30 foot) wells. In deep wells the slotted distance was 6 feet to allow for measurement of groundwater fluctuations.
2. Silica sand was poured down the auger and the auger slowly raised 2 to 4 feet, allowing the sand to filter through the auger and into the well. This procedure was continued until the sand covered the slotted portion of the PVC tubing.
3. The auger was pulled another 2 to 4 feet as bentonite in pellet or powder form was poured down the auger. Enough bentonite was added to yield a 3 foot plug over the sand.
4. The auger was then pulled completely from the well. This usually resulted in 1 to 4 foot of caving of the natural materials.
5. At the end of each day, all wells installed that day were backfilled to within 2 feet of grade with a cement and bentonite grout poured from a mixer truck. A 2 1/2-foot galvanized pipe was then placed around the PVC tubing and the well was brought to grade with concrete (sacrete). Both the PVC tubing and galvanized pipe were capped.

The subsurface conditions encountered during the installation of gas monitoring wells at Mississauga Landfill were somewhat variable. Subsoils encountered within the exploratory borings consisted generally of interlensed deposits of silty and gravelly sands. These granular soils extended to the maximum depth explored of 55 feet. Results of penetration resistance tests indicate the granular soils may be classified as medium dense to dense. Groundwater was encountered at depths on the order of 50 feet. Boreholes 1-80, 2-80, and 3-80, located within the landfill, encountered approximately 5 feet of fill underlain by refuse. The refuse was approximately 47, 14 and 5 feet thick in Boreholes 1-80, 2-80, and 3-80; respectively. Granular soils were encountered below the refuse in each hole.

On September 22, 1980 the gas interceptor system at the Mississauga site was turned off. Personnel from OSU and Morrison Beatty have been monitoring the movement of methane on a weekly basis since September. This simulation is scheduled to end in mid-December. Results of the simulation will be reported at a later date.

Gas Sampling

In accordance with the proposal for this study, the soil atmosphere at the landfill sites is being sampled on a periodic basis. The samples obtained are returned to the OSU Laboratory for gas chromatograph analysis. The results of the chromatograph analyses will be used to verify and/or calibrate the field explosimeter readings of methane concentration.

Computer Simulation of Methane Migration

The analytical portion of this study consists primarily of computer model simulation of methane migration in the soil surrounding each of the subject landfills. The input required for the simulation includes subsurface soil conditions, soil properties, and the geometric configuration of the landfill and surrounding soil. During the Autumn of 1980, the geometry of the Mississauga site was used in a parametric study of the computer

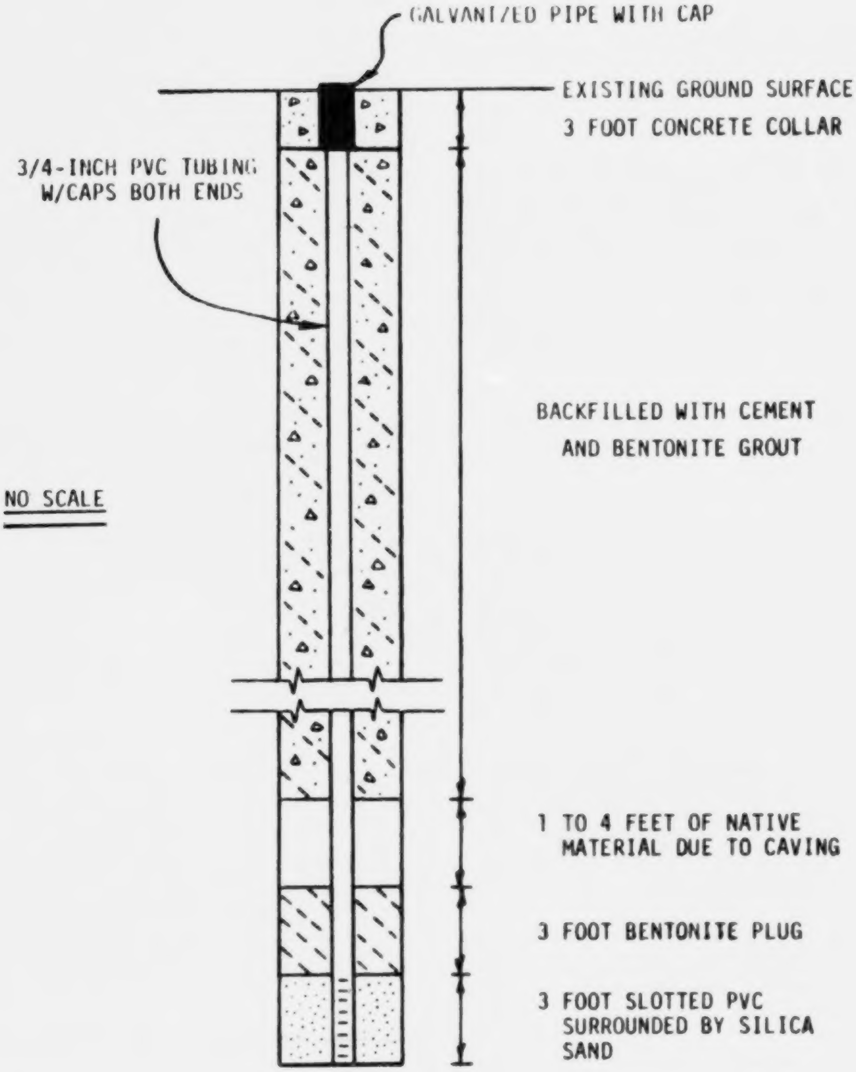
GALVANIZED PIPE WITH CAP
EXISTING GROUND SURFACE
3 FOOT CONCRETE COLLAR
3/4-INCH PVC TUBING W/CAPS BOTH ENDS
BACKFILLED WITH CEMENT AND BENTONITE GROUT
NO SCALE
1 TO 4 FEET OF NATIVE MATERIAL DUE TO CAVING
3 FOOT BENTONITE PLUG
3 FOOT SLOTTED PVC SURROUNDED BY SILICA SAND

Figure 5. Typical Construction of a Methane Monitoring Well

model for diffusional flow. The results of this study will be reported at a later time.

Summary and Conclusions

To date (December 1980) field instrumentation has been installed at two field sites and initial computer simulation studies have been completed. Currently, data are being acquired at the two field sites, and the verification of the computer simulation is being accomplished.

The work remaining to be done on Phase III of the contract consists of the selection of a third verification site. The site selection and implementation of Phase IV, the verification of control systems, is in its initial stages.

PLANTING TREES AND SHRUBS IN LANDFILL COVER SOIL

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ABSTRACT

During the past six years, vegetation growth on completed refuse landfills has been investigated through a nation-wide mail survey, follow-up field visits and carefully monitored field and greenhouse research. Soil gas contamination, surface settlement, soil compaction, thin, poor quality cover soil, low moisture, high soil temperatures and toxic components in the soil matrix have adversly affected tree and shrub growth on landfills. Vegetative reclamation projects were successful only when these limitations were overcome. No species were tolerant of high landfill gas concentrations ($CO2 > 25\%$, $CH4 > 30\%$). However, some species such as Japanese black pine (Pinus thunbergii) were more tolerant of the landfill than others. Shallow rooted species grew better than deep rooters. Small planting stock (< 1m tall) produced more leaf and stem tissue than large trees (> 2m tall). Constructing barriers to gas migration significantly improved plant growth.

Introduction

Sanitary landfill has been demonstrated to be the least expensive environmentally acceptable means of waste disposal available to date, purportedly possessing the attributes of neatness and safety in addition to relatively low cost. Whereas such sites may have originally been located at considerable distances from residential areas, rapid urban and suburban development in the United States has caused many once remote dumping grounds to come within developed areas. As such they provide an attractive source of much needed land for many purposes. Although conversion to recreational areas or other non-structural usage has long been considered an acceptable end for completed landfill sites, the urgent need for space and for increased tax revenues has caused many municipalities to eye completed landfills for commercial use as well. In rural areas, intensifying land use has resulted in attempts to use completed landfills for growing commercial crops.

The composition of landfill refuse varies considerably depending on its origin be it municipal, industrial, incineration material or sewage sludge. The organic content of solid waste collected from homes, schools, commercial establishments and industries generally ranges from 50 to 75% on a weight basis. Most of these organics are biodegradable and can be broken down into simpler compounds by both aerobic and anaerobic micro-organisms. The rate at which this occurs in a landfill is reported to be a function of (a) permeability of cover material (b) depth of garbage (c) amount of rainfall (d) moisture content of the refuse (e) putrescibility of the refuse (f) compaction (g) pH (h) age and (i) presence or absence of toxic materials.

When the refuse is initially deposited in the landfill, there is enough oxygen present to support a population of aerobic bacteria. This stage lasts from one day to many months (6). The literature indicates $CO2$, and $H2O$ to be the principle products formed in aerobic decomposition (5). The depletion of soil oxygen results in a decrease in the aerobic and an increase in the anaerobic population. During anaerobic decompostion, the possibility exists for production of a wide range of gases and liquids. However, the literature indicates CO_2, H_2, CH_4, H_2S, and N_2 to be the predominant gases with CO_2 and CH_4 making up the largest portion of the soil gas atmosphere (4). These gases can migrate into the cover soil over refuse and displace the O_2 normally present in the soil air spaces.

There has been a considerable amount of work done concerning the effects of excess CO_2 in the root zone on different plant species. In 1914, Noyes saturated soil around tomato and corn plants with CO_2 (12). Both species died within two weeks, but there was no irreversible damage to the soil. A good deal of variation in tolerance to CO_2 between species has been found. Cotton seedlings grown in hydroponic solutions (10) were able to exhibit optimum growth with 10% CO_2 present, provided at least 7.5% O_2 was also present. Thirty to 40% CO_2 in the root zone of cotton seedlings was found to severely reduce root growth in hydroponic solutions. Red and black raspberry (13) were killed when their roots were exposed to 10% CO_2.

Chang and Loomis in 1945 (3) conducted a survey of the literature and found that although some plants could survive O_2 concentrations in the root zone as low as 1 to 2%, most plants would function normally at O_2 concentrations ranging from 5 to 10%. There is also a good deal of variability in tolerance to low O_2 in the root zone among different species of plants. One-tenth percent O_2 in the flooded root zone of apple trees resulted in the death of the trees (2). Tomato plants grown in solution culture exhibited marked reduction in growth and ability to take up potassium when exposed to 3% O_2 in the root zone (15). Sour-orange seedlings in sand-solution culture given 1.5% O_2 in the root zone for seventeen weeks did not grow, and seedlings receiving 4.6 to 5.1% O_2 grew half as well as the controls (9). Rice plants have been reported to grow as well in solution culture having less than 1% O_2 in the root zone as control plants (15).

Landfills are being converted into usable recreation areas at a very rapid rate. Since this trend is likely to continue for many years because of the prevalence of landfills and the need for more urban space, we will continually learn more about how to overcome problems associated with growing plant material on landfills. We should not be satisfied with the information contained in this report, for many questions remain unanswered. However, these pages will set the ground work for future investigations and provide a guideline for vegetating completed landfill sites at the present time.

Objectives

This study was designed to investigate the problems associated with growing woody vegetation on completed refuse landfill sites. The objectives of these investigations are to:

1. Determine the extent of vegetation growth problems throughout the U.S.
2. Determine why certain reclamation efforts have been successful and others unsuccessful.
3. Screen tree and shrub species for tolerance to landfill conditions.
4. Identify the factors which limit vegetation growth
5. Determine methods for overcoming the factors limiting plant growth.
6. Draw a set of recommendations for planting treses and shrubs on completed landfills.

Evaluation of Plant Growth on Landfills Throughout the United States

A mail survey of about 1,000 individuals, presumed to be knowledgeable of the vegetation associated with operating the completed landfills throughout the continental United States and territories, was conducted for the purpose of determining the status of landfill vegetation growth. Of the 500 people responding, approximately 75% were unaware of any problems or reported none. Twenty-five percent reported problems on landfills and seven percent reported problems with vegetation adjacent to landfills.

Using reports received through the mail survey, landfills for site visits were selected to represent the nine major United States climatic regions as defined by Trewarthsa (14). About 60 individual landfills were visited, and comparisons of the quality of soil atmospheres were made in the root zone of healthy specimens and individuals of the same species that were dead or dying. Almost invariably, where soil atmospheres contained high concentrations of landfill gases, vegetation was adversely affected. It was also found that in 40% of the cases, there was a discrepancy between conditions reported in the mail survey and those found in on-site visits to the landfills.

A wide variety of landfill reclamation projects were found throughout the country. They ranged from no program at all but where volunteer vegetation had colonized the site to a very well managed botanical garden containing over 2,000 plant species. Trees and shrubs were planted on some dumpsites and never irrigated; these projects failed. Others were more successful because they were managed on a permanent or part-time basis. One key to success certainly included establishing a management program having provisions for irrigation, fertilization, liming and pest control.

Little variability in the magnitude of landfill gas production and consequent vegetation damage was observed among the different climatic regions, except for the arid area (southwestern Arizona) where concentrations of combustible gas and carbon dioxide were found to be somewhat less than in the eight other regions. This was presumably due to lack of rainfall.

Soil characteristics observed to increase in landfill gased soils, include content of moisture and available ammonium-nitrogen, iron, manganese, zinc, and copper. Increased availability of these elements is believed to be due to the highly reduced conditions in the soils and the activity of anaerobic microorganisms. Soil pH was found to approach neutrality in gassed soils. This was probably due to the presence of organic acids produced during anaerobic decomposition of the buried refuse.

Factors Affecting Plant Growth on Landfills and Their Possible Correction

Gas

A survey of soil atmospheres on twenty completed sanitary landfills throughout the United States revealed that combustible gas (methane), CO2 and O2 readings were concentrated in low and extremely high percentage categories rather than being evenly distributed among all of the categories (7). The combustible gas readings were most extreme in this respect, with 86.2% of the samples containing less than 10% combustible gas by volume, whereas, 12.3% of the samples contained 25% or more combustible gas and only 1.4% of the samples had combustible gas concentrations between 10 and 24.9%. This bimodel distribution is probably due to the tendency of refuse-generated gas to well up in specific areas rather than uniformly over the entire landfill site. This could be due to the fact that certain areas on the landfill are less restrictive of gas flow and act as chimneys for gas release, or to some characteristic of the refuse.

This tendency of gas to occur in isolated areas in the cover material could be useful when vegetating these sites. By locating the areas where the gases are present, high concentrations can be avoided and the loss of expensive trees and shrubs can be minimized.

Cover Soil Thickness

The amount and quality of soil covering the refuse has frequently been found to be inadequate for vegetation growth, probably because of costs and availability. Such deficiencies need to be corrected in areas where you wish to grow deep rooted woody vegetation. Generally, shallow soil areas support poor vegetation growth. Because of the excessive cost of covering an entire landfill with deep, rich soil, it is suggested that consideration be given to mounding soil in areas where deeper rooted woody vegetation is to be planted. If no measures have been taken to prevent gas migration from the refuse into the final cover soil, the placement of a thick (0.3 m) clay barrier beneath the soil mound should be considered. These methods will be described in detail subsequently.

Cover Soil Quality

Soil used for cover is frequently below recommended minimum standards for plant growth. It is often low in moisture and nutrient content and may have the wrong pH. These problems must be overcome in order to stimulate good plant growth.

Toxic Compounds in Cover Soil

Tests of soils containing high concentrations of anaerobic gases frequently reveal abnormally high contents of ammonium-nitrogen and manganese. The content of heavy metals such as iron and zinc may also be considerably higher than normal in soils which have been anaerobic for an extended period of time. Elevated levels of these soil compounds can cause toxic symptoms on many plant species. These symptoms can include stunted growth and death.

Soil Compaction

Cover soils frequently become highly compacted during the closing operations of a landfill site due to the use of large heavy earth moving equipment. High soil compaction has been shown to adversely affect growth of trees and shrubs on completed landfills and other areas . Compaction is also a problem in farm fields, where tractor compacted soil can inhibit root development. For planting in highly compacted soil, bulk densities above 1.7-1.8 g/cc, several provisions can be used which will promote better root growth. These include disking organic matter into the soil, adding gypsum or rototilling on small areas.

Settlement

Refuse and soil settlement will create an undulating surface frequently causing water to accumulate in low areas during rainy and irrigation periods. Trees, shrubs and grass in low areas may eventually die if water remains for extended periods and/or rainfall is frequent. At the same time it may not be possible to adequately irrigate the elevated areas. Undulating greens will not be tolerated in most golf courses and, therefore, provisions should be made to prevent green settlement. Operation of farm equipment may be hindered in settled areas of cropland. On the other hand, in extremely dry climates, plant growth may be greatly enhanced in the settled areas due to increased soil moisture in these areas.

Soil Moisture

The nature of the landfill soil strata (i.e. consisting of a meter or less of soil cover over many meters of refuse) and, perhaps, the higher soil temperatures generally found on landfills help promote drying of the cover soil. Higher rates of leaf transpiration observed on trees growing in landfill soil may also contribute to severe soil desiccation. Since moisture content is significantly lower in landfill soil, plants on landfills will require a greater amount of irrigation in order to exhibit growth comparable to non-landfill areas.

Soil Temperatures

Temperatures in anaerobic soils are frequently higher (from a degree or two to perhaps 20 or 30 C in extreme cases) than nearby aerobic soils. The reason for these high soil temperatures is not clear. High soil temperatures have been detected on occasion around dead trees, shrubs and grass and evidently they have an effect on growth in landfills. However, this problem is generally little cause for concern compared to the previously mentioned more frequent occurring limiting factors.

Screening Species for Landfill Tolerance

Gas Tolerance

One desirable characteristic of the plant material would appear to be tolerance to elevated levels of carbon dioxide and methane and low levels of oxygen. Unfortunately, our research and the literature indicates that there does not appear to be any species which can tolerate high levels of these gases (e.g. CO_2 25%, CH_4 30%). However, there are several other criteria, identified during these investigations which will greatly enhance vegetation growth.

Rooting Depth

Tree and shrub species which enjoy a shallow root system were found to be significantly more adaptable to landfill sites than species requiring a much deeper root system (Table 1). The deeper roots are subjected to higher concentrations of landfill gases and lower concentrations of O_2. Some species can avoid this gas environment better than others by producing a shallow root system. Observations at the South Coast Botanical Garden on a former 87-acre landfill site in Palos Verdes, California, showed that shallow-rooted plants seldom are affected by landfill gases, but on some occasions there has been root damage to the deeper rooted larger trees and shrubs (7).

TABLE 1. MEAN ROOT DEPTH* FOR SEVERAL SPECIES ON LANDFILL AND CONTROL PLOTS

Species#	Landfill	Control
	-------cm------	
Japanese Black Pine	7.8	9.3
Norway Spruce	5.1	4.2
Hybrid Poplar (rooted cuttings)	6.3	13.6
Honey Locust	8.3	16.6
Green Ash	9.3	14.7
Hybrid Poplar (saplings)	8.5	12.8

*Each value is the mean of 2 replicates.

#Species are arranged from most tolerant to least tolerant of landfill conditions.

This emphasizes the need for more frequent irrigation in landfill soils planted with woody vegetation than comparable non-landfill areas. In our studies at the Edgeboro Landfill in New Jersey, five of six species excavated for extensive root studies on the landfill plot produced a significantly larger portion of their root system in the top soil layers than trees growing in the nearby non-landfill control area. Figure 1 shows that hybrid poplar growing on the landfill plot has very shallow roots, whereas, figure 2, shows the much deeper root system in the non-landfill area. Because many roots were growing so close to the surface and the top several centimeters of soil regularly dry out for extended periods in the temperate zone of North America, landfill soils must be irrigated more frequently than nearby non-landfill areas to ensure good vegetation growth. Table 2 shows that sugar maples grew significantly better on an irrigated plot than on a non-irrigated area at the Edgeboro Landfill.

Flood Tolerance

In greenhouse lysimeters, red maple, a flood tolerant species was found to be more tolerant also of soil conditions simulating landfill gas than sugar maple, which is sensitive to flooding, if the trees were irrigated regularly. This suggests that flood tolerant species may grow well on landfills because of their ability to withstand low soil O2 cover, provided that they are adequately irrigated. In field experiments on the Edgeboro experimental landfill plot, flood tolerant species (i.e. green ash, honey locust, American sycamore, red maple) grew very poorly on the landfill compared to species not tolerant of saturated soils. The soil was significantly dryer on the landfill plot than on a nearby non-landfill control. Perhaps the former species would have fared better on the dump site if they had been supplied with additional irrigation. Much more research is needed to state with confidence that flood tolerant species will withstand landfill conditions if supplied with enough water.

TABLE 2. MEAN SHOOT LENGTH* OF SUGAR MAPLES IN LANDFILL AND CONTROL IRRIGATED AND NON-IRRIGATED AREAS

Area#	Landfill	Control
	-------cm-------	
Irrigated	20.1b+	24.6c
Non-irrigated	15.4a	23.2c

*Each value was computed from measurements on 30 trees.

#Thirty 2-year old seedlings were planted in each of 4 areas: Landfill and control irrigated and non-irrigated areas.

+Values followed by different letters are significantly different at $p<.01$.

Growth Rate

There is evidence that slow growing trees are more tolerant to landfill conditions than rapidly growing species. Rapid growers generally draw more moisture from the soil than slow growers and would, therefore, require more irrigation than the latter in order to maintain growth comparable to that on a non-landfill area. However, if one is not concerned about the relative growth on the landfill compared to a non-landfill area, then a faster growing tree may be more desirable, to provide a more quickly produced vegetative cover. It has been demonstrated in our studies that species classified as fast growers produce more absolute growth on a landfill than the

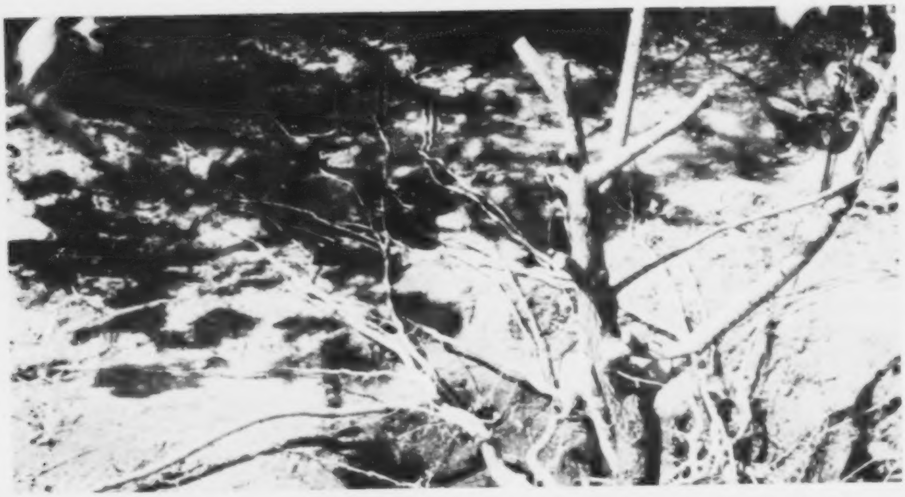

Figure 1. Shallow root system of hybrid poplar growing on the landfill plot.

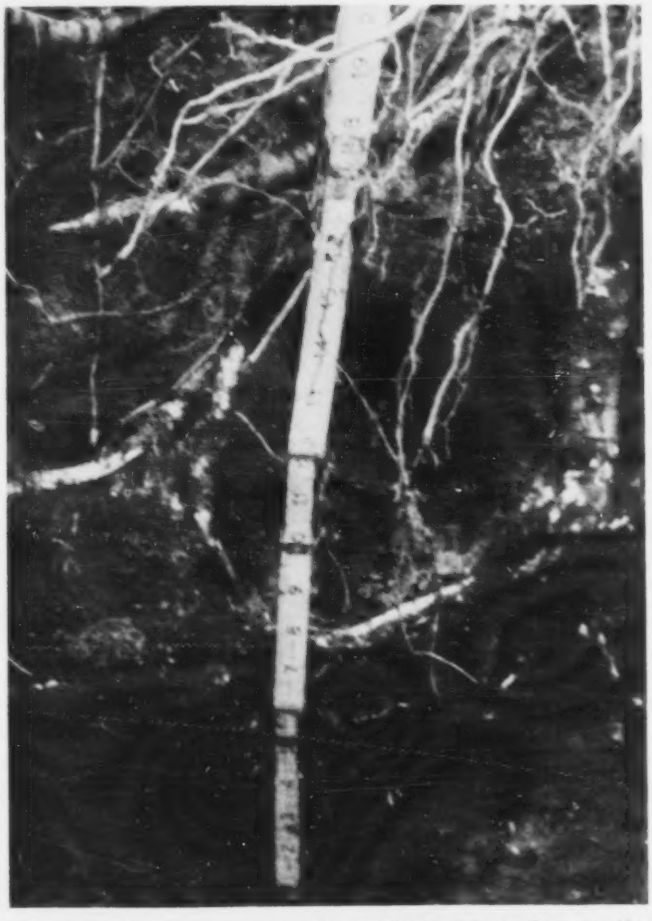

Figure 2. Portion of the deep root system of hybrid poplar on the control plot.

slow growers if they are regularly irrigated during the first three years after planting.

Size at Maturity

Another factor to consider when selecting species for landfills is size of the tree at maturity. If the cover soil is relatively shallow (0.5 m) then it is best to plant a tree which remains relatively small at maturity, or risk tree toppling during high velocity wind storms. If a deeper soil cover is used (1 m or more), then the risk of windthrow will be diminished because the root system has adequate soil space to produce anchor roots, provided, landfill gases are kept out of the cover soil and the trees have adequate moisture.

Constructing Barriers for Preventing Contamination of the Root Zone

The migration of decompositional gases into the root zone of grass, shrubs and trees will adversely affect their survivability. It appears that probably the most expensive but effective procedure available to increase the changes of adequate vegetation growth on former sanitary landfills is by active extraction of gases from the refuse layers. Conceptually, this reduces the chances for gas contamination of the cover soil where vegetation is to be planted. If this procedure is not practical, you should consider the placement of gas barriers between the refuse and the roots to prevent migration of gases into the root zone.

Trenches

Placing a thick (0.3 m) layer of impermeable clay over the final layer followed by an adequate amount (0.6 m or greater) of fertile soil has been shown to be effective in preventing the upward migration of landfill gases into the root zone of woody trees and shrubs. This technique has functioned successfully both on a large scale in California (an area covering approximately one acre)and on a smaller scale in New Jersey. A (0.3 m) layer of gravel placed over the refuse overlain with plastic sheeting and vented with vertical PVC pipes (Figure 3) has also apparently successfully prevented gas contamination of tree roots growing in soil above this barrier. Only one replicate of each method was included in the experiments. Future studies should include two or more replicates.

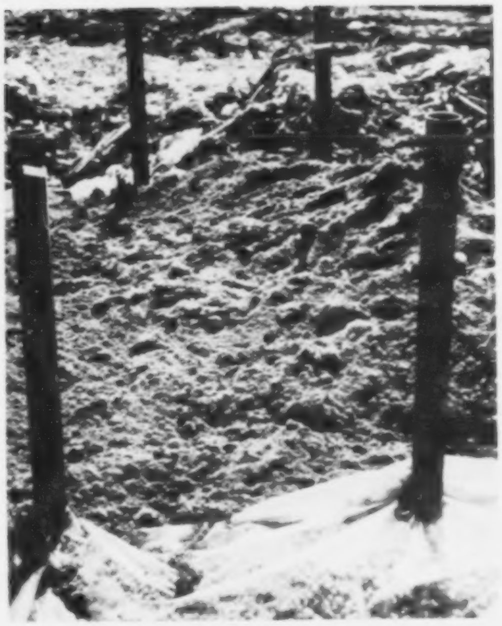

Figure 3. Gas barrier system at bottom of trench showing gravel, plastic sheet and vent pipes.

Mounds

An alternate method of excluding landfill gases may consist of mounding soil only in areas where deeper rooted vegetation (trees and shrubs) is to be planted. The mound may be underlain with a clay layer or plastic (at least 10 mil) to prevent upward migration of decompositional gases. Mounding has been successful in California (Figure 4) and in New Jersey

Another method, although untested, is construction of a saucer lined with clay or plastic (or some other relatively gas impervious material) with a "U" tube installed at the bottom of the saucer (Figure 6). The "U" tube would allow accumulated water to drain out the bottom while preventing gases from backflushing into the saucer. The "U" tube-liner junction must be made gastight.

Proper functioning of all the aforementioned gas-barrier techniques relys

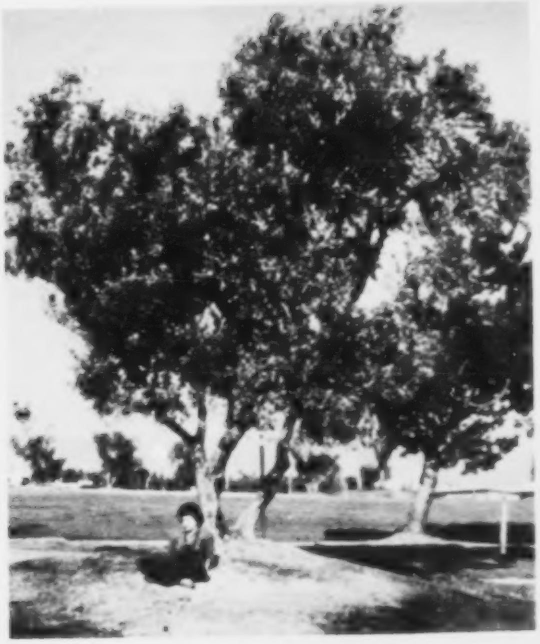

Figure 4. Acacia tree growing in a 1 m high soil mound on the Alemeda Golf Course constructed over a completed landfill.

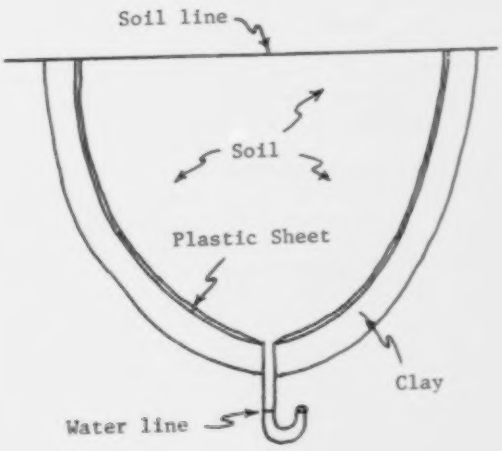

Figure 5. Saucer-lined barrier system

on the assumption that the gas seal will remain intact. Since refuse decomposition will probably continue for many scores of years, refuse settlement is likely to also continue for some time. Cracks or breaks in the gas barrier seal resulting from such settlement may allow decompositional gases to migrate through the barrier. Despite the apparent obstacles, these systems are likely to greatly increase chances of successful vegetation establishment.

Size of Planting Stock

Trees planted when small (<1 m tall) showed significantly better growth on the landfill than those which were planted when larger than 2 m tall, regardless of species (Table 3). Our data have shown that this is related to the ability of a small tree to adapt its root system to the adverse environment in the cover soil by producing roots closer to the surface (i.e. away from the higher landfill gas concentrations deeper in the soil), whereas, roots of larger trees start much deeper and cannot grow to the surface before being killed by the gases. Our data shows that, in fact, by the time the large hybrid poplar saplings adjusted to the landfill by producing a shallow root system, the smaller specimens which started with a shallow root system had grown to a larger size than the saplings.

Effects of Mycorrhizae Inoculation on Tree Growth

Mycorrhizae fungi, when in association with plant roots, have been shown to greatly increase water and nutrient uptake (11). This symbiotic association has been successfully used in coal strip mine reclamation. Mycorrhizae may also aid in successfully establishing vegetation on completed dumpsites since landfill cover soil frequently has a poor capacity for holding nutrients and water. Spore and mycelium inoculated soil has been tested for its ability to promote mycelium development on trees in landfill cover soil. Our results indicate that both forms of inoculum may be viable alternatives to planting trees in un-inoculated soil. The spore inoculum appears to be more suited for higher gassed areas. The reasons for this are unclear at the present time. The roots may be inoculated directly just before planting, thus increasing the likelihood of successfully establishing the beneficial mycorrhizae relationship.

Landfill Gas Effects on Tomato Plants

Carbon dioxide contamination in the root zone of tomato plants in solution

TABLE 3. SHOOT GROWTH ON SMALL AND LARGE SPECIMENS OF FIVE SPECIES ON LANDFILL AND CONTROL PLOTS

Species	Size	Landfill as % control
Pin oak	Small	76.2*
	Large	63.0
Green ash	Small	67.8
	Large	44.3
Honey locust	Small	45.0
	Large	81.4
Sugar Maple	Small	68.4
	Large	45.5
Hybrid poplar	Small	92.8
	Large	39.5

* Calculated from at least five trees per plot.

culture was toxic when the concentration of CO2 averaged 17.0% during the experimental period. When CO2 concentrations averaged 8.8% or less no symptoms were observed, indicating that there was a threshold level between 9 and 17% at which CO2 became toxic under these experimental conditions.

Exposing the roots of tomato plants to 43% CH4 for an 8-day period resulted in no measurable adverse effects, whereas, a 12-day exposure resulted in a decline of the tomato plants concomitant with a decrease in O2 concentration in the culture vessels.

No interaction was observed among O2, CO2 and CH4 when they occurred together in the root zone in terms of symptom development on tomato plants. When CO2 was held at 27%, O2 at either 5.5 or 16% caused similar symptom development. Plants exposed to 34 to 38% CO2 alone exhibited the same symptom development as plants exposed to 34 to 38% CO2 with 43% CH4.

Recommendations

1. Consult the most recent publications on vegetating dump sites before starting a planting program.
2. Choose a soil with a high clay content for the primary cover material to help prevent gas migration into the root zone. A 0.7 m thick layer of loamy textured soil spread over this 0.3 m thick clay layer is likely to promote good root growth and provide for a successful reclamation project.
3. The clay layer functions best if the clay is highly compacted. On the other hand, one should avoid compacting the soil which will support the roots of trees and shrubs (i.e. the top 0.5 m of cover material).
4. Consider discing organic matter into the cover soil if compaction reaches 1.6 g/cc or greater.
5. Avoid soils which are in the anaerobic condition (i.e. soils which contain no oxygen, have a high CO2 and CH4 content and smell very putrid).
6. Check at least the nitrogen, potassium, phosphorus, manganese, iron, copper and zinc levels in the cover soil before planting. Amend the soil according to soil test results. Consider adding lime or sulfur to the soil if the pH is lower than 5.5 or higher than 7.0.
7. Select plant species which have a naturally shallow root system, are volunteer species, have the ability to develop a mycorrhizae relationship or do not attain large size (>15 m) at maturity.
8. Plant trees when they are as young as possible, preferably 2-3 years old.
9. Avoid planting highly disease susceptible vegetation.
10. Construct barriers to vertical gas migration in areas where valuable trees and shrubs will be planted. These barriers may consist of plastic sheeting (at least 10 mil preferably thicker), clay (at least 0.3 m thick) or other relatively gas impermeable material.

11. Extracting gas from the refuse may help prevent upward migration into the root zone, thus increasing the chances of successfully vegetating the landfill.

12. Irrigate plants on a former landfill more frequently than on non-landfill areas.

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COLLECTION OF REPRESENTATIVE WATER QUALITY DATA FROM MONITORING WELLS

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ABSTRACT

Monitoring wells at six sites of widely varying geologic and hydrologic conditions were selected to study monitoring well sampling and preservation techniques. The goal was to develop a sampling protocol that would yield water samples that are representative of the aquifer being sampled. The effects of four pumping mechanisms, time and rate of pumping, well flushing, and preservation techniques on the water quality of the collected samples were studied. Pumping tests and multiple sampling experiments provided data on which recommended sampling protocols and sample preparation, preservation, and storage procedures are based.

The size and accessibility of the individual well, its hydrologic and chemical character, the chemical constituents of interest, and the purpose for monitoring all affect the selections of the pumping mechanism and procedures for sample collection. Chemical parameters, including pH, Fe, and Zn, are more sensitive to the above variables than other parameters.

Recommended sampling procedures are: (1) conduct a pump test to determine hydrologic properties on which to base sampling frequencies, and time and rate of pumping prior to sampling; (2) use a peristaltic or submersible diaphragm type pump when site location permits; (3) use a bailer in more remotely located well sites; (4) measure pH, specific conductance, oxidation-reduction potential, and alkalinity at the time of sample collection; (5) make field measurements in a closed cell that does not expose the sample to atmospheric conditions; (6) filter samples through a 0.45 μm pore sized membrane immediately after collection; and (7) store and preserve samples according to the U.S. EPA (1979) recommended protocols.

INTRODUCTION

The use of wells or piezometers for collecting water samples and water level data has been, and probably will continue to be, the traditional method for monitoring the effects of waste disposal facilities on groundwater. Water sample collection and preservation techniques have been established by several different laboratories and agencies in an attempt to insure that water samples delivered to the laboratory are chemically representative of the water sampled. However, there is considerable controversy between laboratories, agency policies, and researchers concerning proper techniques of sampling from monitoring wells and the appropriate procedures for preserving the original chemical character of the samples. If monitoring wells and water samples are to provide the performance yardstick of disposal facilities design and operation, the significance of the various sampling procedures and preservation techniques must be determined.

The three principal purposes of this study were: 1) to determine if current sampling methods produce samples that are representative of water contained in the aquifer being monitored; 2) to determine if groundwater samples collected in the field must be treated (filtered and acidified) on location, or if they can be brought back to the laboratory for treatment without altering their water quality; and 3) to determine which sampling and preservation techniques should be accepted as standards for monitoring well sampling. In order to fulfill these objectives, the following specific goals were set:

1. Determine the hydrologic properties of the materials tapped by each monitoring well studied.
2. Determine a pumping scheme for each well to obtain water samples representative of aquifer water.
3. Collect a series of samples from each well using four different pumping methods.
4. Determine the effects that the pumping mechanism, time and rate of pumping, and preservation techniques have on the water quality of samples collected.
5. Recommend monitoring well sampling procedures and sample preservation techniques for specific chemical constituents.

Because of the enormous quantity of data produced by the above scheme, certain sampling practices had to be accepted. All of the samples were preserved according to the U.S. EPA's (5) recommended procedures. Also, all of the samples were stored in linear-polyethylene bottles. These practices were well documented in the literature and are not detrimental to a sample's integrity for short time periods. During the collection, preparation, preservation, and storage of the samples, no new procedures or equipment were used.

METHODS

Site Selection and Descriptions

Monitoring wells at six sites in the State of Illinois were selected for study. These sites represent a wide variety in hydrologic conditions and aquifer chemical characteristics. Two sites are active sanitary landfills; two are inactive sanitary landfills; one is a secondary zinc smelter; and one is a disposal site for a hog processing plant. The wells were installed prior to this study. Five are cased with PVC pipe, either 3.81 or 5.08 cm (1½ or 2 in.) in diameter. The sixth well is cased with 5.08-cm (2-in.) diameter galvanized iron. The well depths range from about 5 to 10 m (16 to 30 ft) and have nonpumping water levels from 0 to 5 m (0 to 16 ft) below land surface.

For this paper, data collected at Site 6 are presented. Site 6 is an inactive landfill located in northeastern Illinois. Filling by the trench and fill method began in September 1952 and was completed by November 1966 [Hughes et al. (1)]. Household and garden refuse was the major component of the fill, but small amounts of spent battery acid, construction debris, and sewage sludge were also buried at the site.

The well chosen for study is MM 63 constructed in 1968 by Hughes et al. (1). The well is 3.81 cm (1½ in.) in diameter and 5.12 m (16.8 ft) deep. It is cased with PVC pipe and has a 30.5-cm (1-ft) screen at depths from 4.82 to 5.12 m (15.8 to 16.8 ft). The well reportedly penetrates the landfill refuse and is completed in a 1.22-m (4-ft) sand layer immediately beneath the refuse.

Pumping Equipment

Four pumping methods were selected for collecting samples from the monitoring wells: a peristaltic pump, an air-lift system, a nitrogen-lift system, and bailing. For conducting pumping tests and collecting samples by mechanical means, two different pumps were used. In wells where pumping lifts were within suction lift capabilities, a Masterflex 7545 variable speed drive unit equipped with a 7015 peristaltic pump head was used. For wells where pumping lifts were beyond suction lift capabilities, a diaphragm type pump modeled after the Middleburg pump was constructed and used. This type of pump is available commercially. The flow rate and lift capability from this type of pump is controlled by varying the frequency of alternately applied and released pressure on the diaphragm and by regulating the pressure at which it operates. The pump constructed for this project was 45.72 cm (18 in.) long, 3.18 cm (1¼ in.) in diameter, and delivered water from 12.19 m (40 ft) at a rate of 3,500 ml/min (.92 gal/min).

The air-lift and nitrogen-lift pumping systems used an apparatus similar to that shown in figure 1. A 1.27-cm (½-in.) diameter rigid PVC discharge pipe and a 0.635-cm (¼-in.) plastic airline were used for some sampling runs. For others a 0.952-cm (3/8-in.) diameter flexible polyvinyl discharge line and 0.635-cm (¼-in.) plastic airline were used. All air-lift sampling runs used a four-cylinder electric driven air compressor. For the operation of the nitrogen-lift pumping mechanism, 8.5 m^3 (300 ft^3) capacity cylinders of compressed nitrogen gas were used to pressurize the identical apparatus used in the air-lift pumping experiments. A 2.54-cm (1-in.) diameter stainless steel bailer, 0.914 m (36 in.) long, was constructed for bailing of samples. This bailer retrieves a sample of about 300 ml (10 fl oz).

To provide a basis for comparison, the vertical point of sample collection from within the well for all pumping mechanisms was the mid-point of the well screen. The pump intakes for the mechanical, air-lift, and nitrogen-lift systems were set in each well at the mid-point of the screen. When bailing, the bailer rope was marked such that the bottom of the bailer would be lowered to the same point in the well (the mid-point of the screen) during each successive bail.

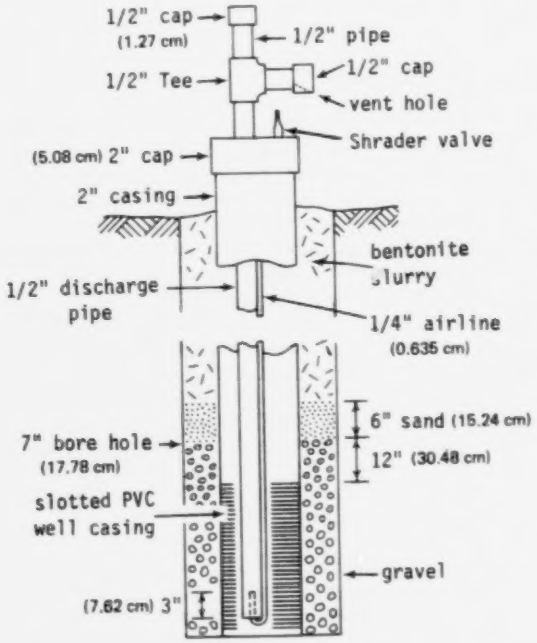

Figure 1. Typical air- and nitrogen-lift pumping mechanism.

Sample Collection and Preparation

Each site was sampled once a month by one of the four pumping mechanisms. This sampling scheme was used to study the effects of the pumping mechanisms on sample quality, the effects of well flushing, and the effects of filtration. The frequency of sampling allowed ample time for the wells to recover from pumping during the previous sample collection. Samples were taken from the initial water in storage, designated as the zero well volume, and at ½, 1, 1½, 2, 4, 6, 8, and 10 well-volume intervals. One well volume is defined as the amount of water occupying the well casing before pumping is initiated.

Measurement of pH, Eh (oxidation-reduction potential), specific conductance, and alkalinity were made on unfiltered aliquots of each of the well volumes or partial well volumes sampled. The sample aliquots were subdivided into three portions by filtering through a 3.0 µm, 0.45 µm, or 0.22 µm pore sized Millipore^R membrane. Satorius^R plastic filter holders fitted for 47 mm diameter filters were used with compressed nitrogen gas overpressure to speed filtration. Each filtered subsample was then further subdivided into samples for total organic carbon (TOC), cation, and anion analysis. All filtering was conducted in the field immediately after each sample was collected.

All samples were stored in linear polyethylene bottles that had been washed with dilute nitric acid and rinsed copiously with distilled water. Detailed procedures used for sample preservation are listed in U.S. Environmental Protection Agency (5).

Chemical Analytical Methods

The Jarrell-Ash^R Atom Comp 975 direct reading emission spectrophotometer (ICP), which has an inductively coupled argon plasma source, was used for the determination of Al, As, B, Ca, Cd, Cr, Co, Fe, Mg, Pb, Se, and Zn. Na and K were determined by atomic absorption while Cl^-, F^-, pH, and Eh were measured electrometrically with specific ion electrodes and a platinum electrode for Eh measurements. Sulfate was determined turbidimetrically. Nitrate analysis was accomplished by the cadmium reduction method and TOC's were measured with an Oceanographic International 0524B total carbon system.

Field measurements of pH and Eh were conducted with an Orion^{R} 407A field model pH meter. Specific conductance (EC) was measured with a YSI^{R} model 33 S-C-T meter and alkalinity was measured by titration to the methyl orange end point.

RESULTS

The proper procedures for collecting water samples from monitoring wells must be selected on the basis of the hydrology of the site, the physical characteristics of the well, the chemistry of the water to be sampled, and the parameters of interest. A comprehensive understanding of the responses of the well and water sample to the choice of pump, selected pumping rate, time of pumping, and appropriate preservation techniques is essential.

Pump Test Analyses

Controlled pumping tests were conducted in an attempt to determine the hydraulic characteristics of the materials the wells were finished in. An understanding of the yield potential of the well and the transmissivity of the "aquifer" being sampled is essential to devising a proper sampling scheme. Without knowledge of these factors, intelligent decisions can not be made concerning the pumping rate or the length of time the well should be pumped prior to collecting a "representative" sample.

Traditional analyses of pump test data use the equations derived by Theis (4) and Jacob (2). One of the basic assumptions made in deriving those equations is that all of the water pumped from a well during the pumping test comes from the aquifer and none from storage within the well. Since this condition is not always fulfilled in practice, particularly for low yielding wells commonly used for monitoring, the Theis and Jacob equations are inappropriate for describing the behavior of water levels during pumping for most monitoring wells.

Papadopulos and Cooper (3) presented an equation describing the discharge from a pumped well which takes into account the volume of water removed from casing storage. The drawdown (s) inside the well is expressed as

$$s = (6.875\ Q/T)\ F(u,\alpha) \qquad (1)$$

where

s = drawdown in the well in meters at time t,
Q = pumping rate in l/sec,
T = transmissivity in m^2/day, and
$F(u,\alpha)$ = well function as defined by Papadopulos and Cooper.

Drawdown values calculated from equation 1 differ significantly from those from the Theis and Jacob equations during the early portion of the pumping test when a relatively high percentage of the discharge comes from casing storage. During the later stages of the pumping test when only a negligible quantity of water is obtained from casing storage, the equations produce equivalent results.

A pumping test was conducted on the Site 6 well on July 24, 1979. The well was pumped with a peristaltic pump at rates from 765 to 620 ml/min for a period of 2 hours. Recovery measurements were taken for 10 minutes after pumping stopped. With the Papadopulos and Cooper method of analysis, an aquifer transmissivity of about 2.48 m^2/day (200 gpd/ft) was determined.

Once the transmissivity of the aquifer tapped by a monitoring well has been determined, the percent of water coming from the aquifer and that from storage can be determined as a function of pumping time. Figure 2 illustrates the percent of water pumped that is coming from aquifers of different transmissivities at a pumping rate of 500 ml/min. As expected, the higher percentages of aquifer water are derived from the higher yielding wells (larger T values). Once a T value of 62.0 m^2/day (5,000 gpd/ft) is reached, larger values of T make little or no difference. Thus, if the yield capability of the well is sufficiently large (T>5,000 gpd/ft), the effects of water from storage become negligible for 5.08-cm (2-in.) diameter wells and can be ignored when establishing a sampling protocol.

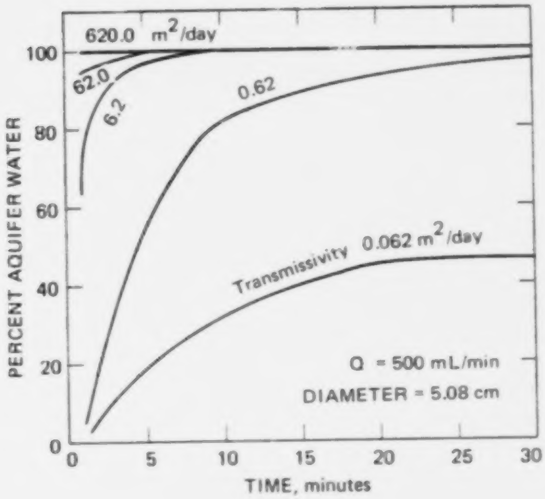

Figure 2. Percent of aquifer water versus time for different transmissivities.

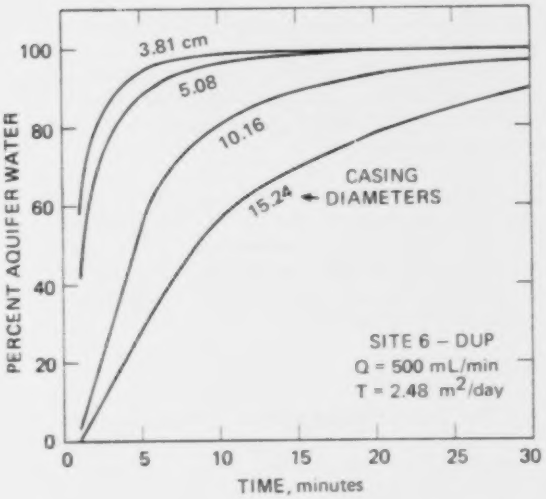

Figure 3. Percent of aquifer water versus time for different well casing diameter.

Additional calculations illustrate the effect of the diameter of the well on the percent of aquifer water removed at a constant pumping rate and transmissivity (see figure 3). For a T of 2.48 m^2/day (200 gpd/ft), well diameters larger than 5.08 cm (2 in.) have a significant effect on the percent of aquifer water pumped, particularly during the very early stages of pumping. Results from analyses of pump test data provide the basic hydrologic information needed to permit a better understanding of the changes in chemical quality of water likely to occur as the sample is pumped from a monitoring well as a function of pumping time.

Effects of Pumping Mechanisms

The pumping mechanisms selected for this study are all commonly used methods for collecting water samples from monitoring wells. The sites were sampled using peristaltic, air-lift, nitrogen-lift, and bailing mechanisms. For discussing the effects of pumping mechanism on sample chemistry, only data for samples filtered through 0.45 µm pore size membranes are presented.

Probably the single most important factor affecting the chemistry of groundwater is pH. Therefore, anything that alters the pH of the groundwater samples is likely to alter the chemistry of its constituents. On-site pH measurements at all six sites using the four pumping mechanisms were analyzed. The data illustrate two possible effects of the type of pumping mechanism on sample chemistry. For comparison, all of the pH data for each site were normalized to the "stabilized" pH values obtained for the 6, 8, and 10 well volume samples collected by the peristaltic pump at that site. The normalized pH values represent the relative differences in pH between samples collected using the peristaltic pump and the other pumping mechanisms. The pH data from samples collected with the peristaltic pump were used as a base because the peristaltic pump works in a closed system and is believed to induce the least amount of sample alteration due to exposure to atmospheric conditions. These delta pH values are plotted in figure 4 for each sample collected.

Data from samples collected using the peristaltic pump at all six sites are grouped at a normalized pH value of 100. The fluctuation in pH of samples collected during the early stages of pumping (0 through 4 volumes) probably was due to the mixture of water stored in the well casing and aquifer water pumped into the well. As a higher percentage of aquifer water was pumped (volumes 6 through 10), the pH values of the samples stabilized.

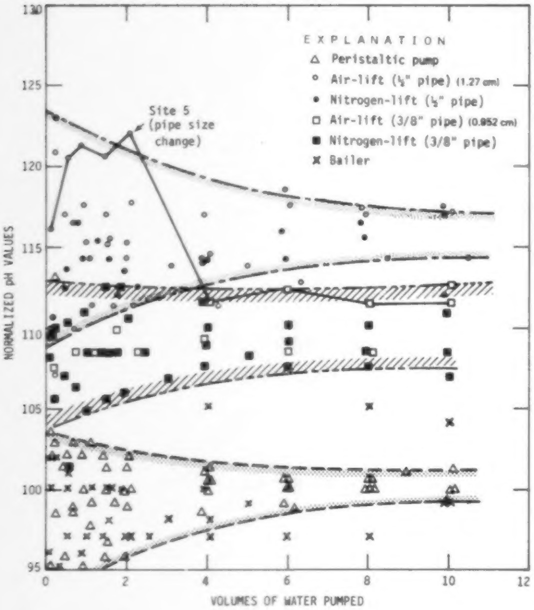

Figure 4. Effects of pumping mechanisms on pH values.

The pH values converged and presumably reflect the pH of water representative of the aquifer.

Two other distinct groupings of data also are shown in figure 4. When pumping the wells with air-lift and nitrogen-lift, two different diameter discharge pipes were used, a 1.27-cm (1-in.) and a 0.952-cm (3/8-in.). Data obtained from samples collected by either air or nitrogen, with a 0.952-cm discharge pipe, appear to stabilize at a normalized pH value of 110. Data obtained from samples collected using either air or nitrogen with the 1.27-cm discharge pipe appear to stabilize at a normalized pH value of 116. In both groupings, larger fluctuations in pH values during the early stages of pumping were noted, presumably due to the mixing of stored and aquifer water, as with the mechanically pumped samples.

Marked pH value changes were noted throughout the duration of sampling with the air- and nitrogen-lift mechanisms. It is postulated that the bubbling air or nitrogen rising through the water was stripping dissolved CO_2 that was in excess of atmospheric pressure from the water. This resulted in increased pH values over those measured in samples pumped with the peristaltic pump. Smaller changes were noted in the samples collected using the 0.952-cm discharge pipe. This probably occurred because less air or nitrogen was needed to pump the same quantity of water than with the 1.27-cm discharge pipe. The gas-to-water ratios for the samples collected using the 0.952-cm diameter pipe varied from about 3.4 to 9.5, with the higher ratios generally resulting in larger pH changes. The gas-to-water ratios for the samples collected using the 1.27-cm diameter pipe were much higher (30 to 40) and appear to have effectively stripped the samples of most of their excess dissolved

TABLE 1. AVERAGE pH VALUES FOR THE SIXTH, EIGHTH, AND TENTH WELL VOLUMES COLLECTED AT EACH SITE BY EACH MECHANISM

Site	Pump	pH
1	Peristaltic pump	4.5
	Air-lift*	5.4
	Nitrogen-lift*	5.3
	Bailer	5.2
2	Peristaltic pump	7.0
	Air-lift*	8.0
	Nitrogen-lift*	8.1
	Bailer	ND†
3	Peristaltic pump	7.1
	Air-lift*	8.2
	Nigrogen-lift‡	7.6
	Bailer	7.3
4	Peristaltic pump	7.2
	Air-lift*	8.2
	Nitrogen-lift‡	7.8
	Bailer	7.0
5	Peristaltic pump	6.8
	Air-lift‡	7.6
	Nitrogen-lift‡	7.5
	Bailer	6.8
6	Peristaltic pump	7.1
	Air-lift‡	7.8
	Nitrogen-lift‡	7.6
	Bailer	7.0

*1.27-cm (½-in.) id inlet line

†No data

‡.952-cm (3/8-in.) id inlet line

CO_2, resulting in pH values of 8.2 to 8.3. This can be compared with pH values of 6.9 to 7.0 when the peristaltic pump was used to collect samples. The measured pH values are listed in table 1.

Data from samples collected at Site 5 using an air-lift pumping mechanism dramatically illustrate the effect of discharge pipe size on pH as shown in figure 4. During the first stage of pumping (volumes 0 through 2), a 1.27-cm discharge pipe was used. To increase the pumping rate, the 1.27-cm pipe was withdrawn and a 0.952-cm pipe inserted between volumes 2 and 4. The immediate drop in pH shown in figure 4 confirms the important effect of the gas-to-water ratios on the pH of water samples collected with gas-lift devices.

Because of the effect of the pumping mechanism on sample pH, it was expected that concentrations of some chemical parameters also would be affected. Figure 5 shows plots of dissolved Fe concentrations determined for each of the partial or full well volumes sampled by each of the pumping mechanisms for Site 6. Iron is extremely susceptible to change in solution concentration due to change in pH or during oxidation-reduction reactions within a groundwater system. The soluble Fe concentrations in the samples taken with an air-lift mechanism remain almost constant through the 10 well volumes (approximately 0.60 mg/l). The initial Fe concentrations in the samples collected with the peristaltic and nitrogen-lift mechanisms were 4.0 and 1.7 mg/l, respectively. However, after pumping 4 well volumes by the three mechanisms, the resultant dissolved Fe concentrations in the samples were nearly identical. This would indicate that the groundwater in the aquifer, volumes 4 through 10, possessed a lower level (approximately 0.60 mg/l) of Fe than the storage water in the well casing. Similar results were noted for Zn.

There was no obvious effect on Ca, K, Mg, Mn, or Na due to the choice of pumping mechanism. There is an absence of any trend which would indicate one mechanism would be better than another when sampling for any of these constituents. The same general patterns of concentration increase or decrease with well volumes pumped occur regardless of the pumping mechanism used. The data should be interpreted with caution because the variation in constituent concentrations in the samples collected by the different pump mechanisms may also result from seasonal fluctuations in groundwater chemistry.

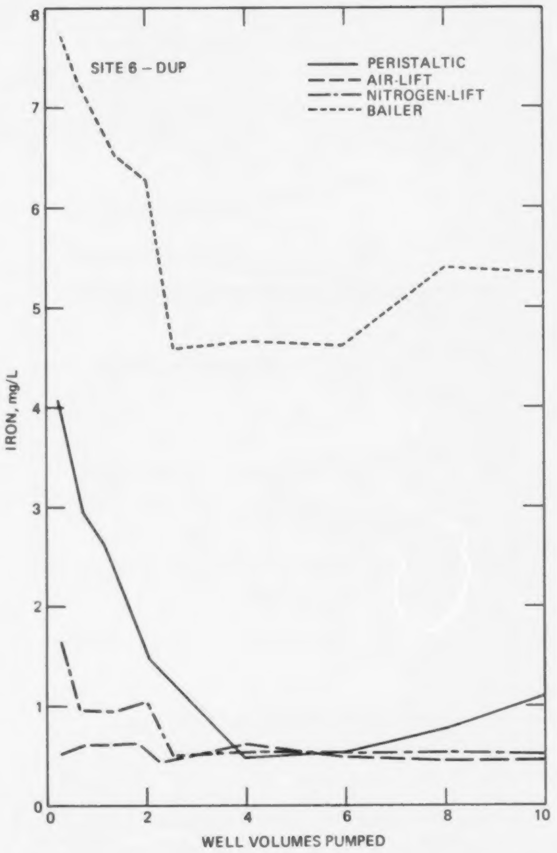

Figure 5. Effects of pumping mechanism on iron concentrations at site 6 as a function of well volumes pumped

To eliminate the possible effects of seasonal variations in site geochemistry, an additional sampling routine was used for Site 6. Ten well volumes were pumped with a peristaltic pump and a sample was taken. The peristaltic pump was quickly withdrawn from the well, replaced by the air-lift system, and another sample was collected. Compressed air was replaced with compressed nitrogen gas and another sample was taken. Finally, the gas-lift mechanism was removed, and a bailer was used and a fourth sample collected. The results were supportive of the conclusion that changes in sample chemistry were induced by the pumping mechanism used and that the differences observed previously were due to changes in groundwater composition resulting from seasonal variations.

Effects of Well Flushing

The primary consideration during the collection of groundwater samples from monitoring wells is to obtain a sample that is representative of the groundwater in the aquifer. The recommended procedure is to flush the monitoring well to remove the stagnant water held in storage in the well casing or to pump until a high percentage of aquifer water is being received. Storage water is defined as the water that doesn't come into contact with the flowing groundwater. Acknowledging the existence of site specific variables of geology, hydrology, transmissivity values, and chemistry between waste disposal sites, the extent of well flushing required will be different for each well. As demonstrated in figure 2, a specific volume of water pumped from one monitoring well before sample collection may be sufficient to produce a representative sample of the aquifer, but at another site the same volume may be insufficient or perhaps result in overpumping. Overpumping may introduce groundwater from a distant source that could dilute or concentrate certain constituents and result in erratic data.

Site 6 possessed the most unique hydrogeologic settings of the six sites. The well was placed directly beneath the refuse at a municipal landfill. The landfill also possesses a relatively thin soil cover which allows for relatively rapid recharge of seasonal variations in groundwater chemistry at the site.

Groundwater within a landfill also has an extreme heterogeneous chemical nature, dependent upon its immediate environment within the landfill. Thus, sampling from a well placed directly into or beneath the refuse could result in samples exhibiting large fluctuations in chemical composition.

The plots of Na, K, Mg, and Ca versus well volumes pumped for site 6 (figure 6) show only slight changes in concentration of these constituents while pumping. Figure 7 depicts the change in Fe concentration during the pumping of 10 well volumes when the well was sampled in both August and February with a peristaltic pump. The decrease in Fe during pumping in August was felt to represent an aquifer under oxidizing conditions where most of the Fe had precipitated. The trend of increasing Fe in the February samples would indicate that the aquifer (or refuse) was under a reduced environment with more Fe in solution. The presence of a frozen soil cover during winter, creating a closed system and not permitting the introduction of oxidants, could explain the reducing environment. The sample collected during August reflect the effects of well flushing (volumes 0 through 4), the oxidizing environment of the summer months (the lower overall Fe concentrations), and the heterogeneous nature of water contained in refuse (volumes 6 through 10). It appears that it would be very difficult to obtain reproducible "representative" samples of water from

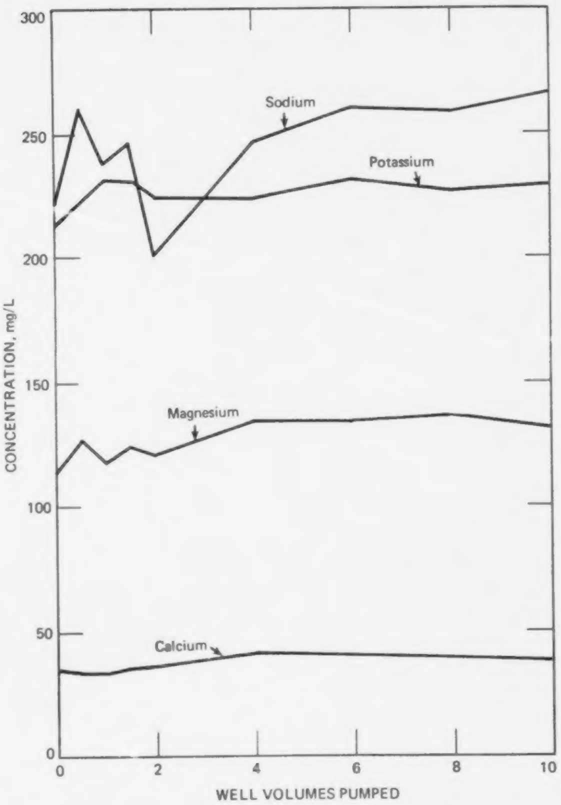

Figure 6. Site 6: Na, K, Mg, and Ca concentrations versus volumes pumped (peristaltic pump).

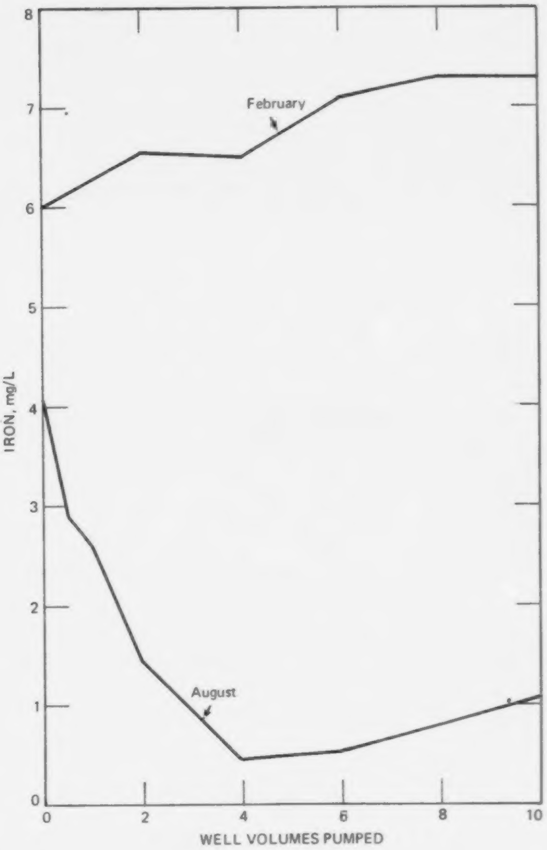

Figure 7. Site 6: Fe concentrations versus volumes pumped for two sampling periods (peristaltic pump).

this monitoring well. There are too many uncontrollable factors affecting the chemical characteristics of water contained in the refuse.

Sulfate, F^-, and Cl^- determinations also were made on samples collected by peristaltic pump. Data show some initial fluctuation in the Cl^- values before a stable concentration was obtained. Overall, the data suggest that the Cl^- concentrations found in the storage waters were similar to that in the aquifers. The effect of pumping on Cl^- concentrations is of concern since Cl^-, along with pH, temperature, and specific conductance, has often been used to indicate when representative groundwater samples have been obtained. The Cl^- values vary only slightly with volumes pumped as do many of the other principal constituents. This suggests that Cl^- would not be a good indicator of when representative aquifer water has been obtained.

The chemical data from this portion of the study verified the theoretical ratios of aquifer to stored water predicted during pumping. One group of elements, Na, K, Mg, Ca, and Cl, is relatively insensitive to time of pumping or well flushing. Another group of elements, Fe, Mg, Zn, Cd, Cu, As, Se, and B, is sensitive to the effects of well flushing. These constituents apparently undergo chemical changes when stored in the well casings. These constituents along with temperature, pH, Eh, and conductivity could be candidates as indicators of when representative samples have been obtained. The selection of the most suitable indicator should be based on the hydrologic and chemical characteristics of each individual site.

Effects of Sample Preparation, Preservation, and Storage

The effort to collect a representative groundwater sample from an aquifer via a monitoring well would be futile if the sample chemistry changed between the time of collection and analysis. Proper sample preparation, preservation, and storage can help prevent such changes from occurring. Adequate procedures have been developed for sample preservation and many investigations into the proper vessels for sample storage have also been completed. This phase of the project was primarily concerned with whether these prescribed procedures for preparation and preservation of samples should be applied on-site immediately after sample collection or in the laboratory some time later.

Four-liter samples taken at the 10th well volume from Site 6 during the February samplings were used for the storage analysis. Calcium, K, Mg, Mn, and Na show little change in chemical composition in response to storage time. The changes in pH, Fe, and Zn, however, were immediate. During the 7 hours between collection of the samples and return to the laboratory, where a subsample of the larger volumes was taken, these parameters underwent significant change. Almost all of the dissolved Fe and Zn present in the samples was lost during the first 7 hours of storage. This loss could be due to adsorption of the particulate matter onto the wall of the storage container. The most probable cause in reduction of the Fe concentrations is precipitation.

The effects of filter pore size on sample chemical composition also were studied. Subsamples, filtered through a 3.0, 0.45, or 0.22 μm pore sized filter, were taken from each well volume or partial well volume collected during the monthly sampling routines. The analyses of these subsamples were then compared to determine the effect of membrane pore size on constituent concentration in the filtered samples.

These data show that for Ca, Mg, Na, K, and Mn the membrane pore size used for sample preparation does not affect their concentration in solution. However, the Fe and Zn concentrations showed a definite effect from filter pore sizes. The highest Fe and Zn concentrations were found in the samples filtered through the 3.0 μm pore sized membrane. The next highest concentrations were found in the samples filtered through the 0.45 μm filters. The lowest concentrations of Fe and Zn were found in samples filtered through the 0.22 μm filters. The samples were very turbid and colloidal material was observed to pass through the 3.0 μm membrane but not the 0.45 or 0.22 μm pore sized membranes. Subsequent acidification of the samples for preservation resulted in leaching or dissolution of Fe and Zn from these colloid particles.

CONCLUSIONS

The results of this study show that collecting "representative" water samples from monitoring wells is not a straightforward or easily accomplished task. Each monitoring well-aquifer system combination has its own individual hydrologic and chemical character that must be considered when planning a sampling protocol. The selection of the type of sampling device; sample preparation, preservation, and storage; and sampling procedures all must be tailored to the size and accessibility of the individual well, its hydrologic and chemical character, the chemical constituents of interest, the time of year, and the purpose for monitoring.

It has been demonstrated that meaningful pump tests can and should be conducted on monitoring wells. The analyses of time-drawdown data from constant rate pump tests using the equations developed by Papadopulos and Cooper (3) yielded realistic transmissivity values for the materials tapped by the monitoring wells studied. Aquifer transmissivities can be used to project time-drawdown relationships at various pumping rates and to determine the percent of "aquifer water" contained in the total amounts pumped at any given time. This relationship can and should be used as a guide for obtaining "representative" water samples from monitoring wells.

The transmissivity values are also invaluable when attempting to determine times of travel of groundwater within the aquifer or materials being monitored. By applying appropriate hydraulic gradients, porosities, and transmissivities, the bulk velocities of groundwater movement can be estimated. These velocities can be used to project the rate of migration of pollutants detected in the monitoring well and to describe the geometry and dynamics of the contaminate plume. The velocity of regional groundwater flow also should be used to determine realistic and economic frequencies for sample collection. In very "tight" materials (low transmissivities) the rates of groundwater movement may be on the order of 0.5 to 2 m per year (1 to 5 ft). If monthly sampling is prescribed in a case such as this, the same "slug" of water may be, in effect, sampled for 2 or 3 months consecutively.

Chemical data from samples collected with the four types of pumps used in this study indicated that peristaltic pumps and bailing yielded comparable data and caused the least changes in chemical quality of water delivered to the surface. Air- and nitrogen-lift pumping mechanisms increased the pH of water samples during pumping and therefore altered the chemical concentrations of pH sensitive constituents. Iron and zinc were shown to be particularly sensitive to the use of these types of pumps.

The effects of flushing out a well, or pumping the well for a period of time to insure collection of a "representative" sample, also have been effectively documented. In most cases, the water stored in the well casing was of different chemical quality than that contained in the aquifer. Usually, the oxidizing environment in the well was sufficiently different from that of the aquifer to create a shift of chemical species in solution.

To insure that a "representative" aquifer sample is collected, the well should be pumped until a high percentage of "aquifer water" is obtained. The length of time of pumping will depend on the rate of pumping, well diameter, and transmissivity of the aquifer being sampled. With these factors, the aquifer water percentages with time can be calculated and used as a guide for determining the appropriate pumping time before a sample should be collected. Monitoring pH while pumping with a peristaltic pump or bailer appears to be a reasonable field check for assuring that a representative aquifer sample is being collected.

Data collected from the same monitoring wells at different times of the year show that significant seasonal variations in chemical quality can occur. These changes can be related to varying rates of recharge and changes in oxidation-reduction conditions in relatively shallow aquifer systems. Awareness of the possibilities of these types of seasonal variations is essential to understand and properly interpret the significance of changes in chemical results with time. Changes in field measured pH or specific conductance values from one sampling period to another should be expected and are not causes to abandon established sampling procedures.

Data from the sample preparation, preservation, and storage portions of the study show that, when chemical concentrations for certain constituents are desired, samples definitely should be filtered in the field at the time of collection. Chemical constituents sensitive to pH changes can be affected within 7 hours if not filtered and preserved at the time of collection. The constituents most sensitive were Fe and Zn.

In addition, the selection of filter pore sizes used to filter samples can affect the chemical analyses results. Mineral constituents associated with clay particles increased significantly when the samples were filtered through 3.0 μm pore size membranes. Samples filtered through 0.45 μ pore size membranes had only slightly higher clay related mineral values over those filtered through 0.22 μm pore size membranes. For practical purposes, the differences in chemical composition were small and the use of the 0.45 μm pore size membrane appears satisfactory. Use of the small pore size filters was more time consuming and resulted in filter clogging problems, particularly for turbid samples.

RECOMMENDATIONS

The purpose or philosophy of a proposed groundwater monitoring program must be agreed upon and used as the basis for locating and designing monitoring wells and establishing applicable sampling procedures. This study was designed to evaluate practical sampling techniques and sample preparation, preservation, and storage procedures. The basic goal was to develop procedures and protocols to obtain chemical results representative of water contained in the materials being sampled.

On the basis of the results of this study, the following recommendations are made.

1) A brief, 2 or 3 hour, pumping test should be conducted on each monitoring well to be sampled. Analyses of the pump test data and other hydrologic information should be used to determine the frequency at which samples will be collected and the rate and period of time each well should be pumped prior to collecting the sample.

2) The general rules of thumb for pumping 4 to 6 well volumes will in most cases produce samples representative of aquifer water. For aquifers with unusually high transmissivities, pumping for periods long enough to remove these volumes of water may induce migration of water from parts of the aquifer remote from the monitoring well. The calculation of percent aquifer water with time provides a more rational basis for determining the length of pumping. Samples should be collected in the minimum time required to produce "representative aquifer water."

3) A controlled sampling experiment, similar to those in this study, preferably using a peristaltic or submersible

diaphragm type pump, should be conducted to accurately determine the chemical quality of the aquifer water and to verify the chemical response of the monitoring well to pumping as predicted from the pump test data. Once the chemical character and responses of the monitoring systems have been determined, key chemical constituents for routine sampling can be selected.

4) On the basis of the sensitivity of the selected "indicators", a choice of pumps for routine sampling can be made. The use of air- or nitrogen-lift pumping mechanisms should be restricted to pH insensitive chemical constituents. Although this study dealt with inorganic constituents, the data suggest that these types of pumping mechanisms probably would also strip volatile organic compounds from the water during pumping. The peristaltic or submersible diaphragm pumps and the bailer are recommended for most applications.

5) The monitoring well should be pumped at a constant rate for a period of time that will result in delivery of at least 95 percent aquifer water. The rate and time of pumping should be determined based on the transmissivity of the aquifer, the well diameter, and the results of the sampling experiment.

6) Measurements of pH, Eh, and specific conductance should be made at the time of sample collection. These measurements should be made within a closed cell which will prevent the sample from coming into contact with the atmosphere. All samples should be promptly filtered through a 0.45 µm pore size membrane and preserved according to recommended U.S. EPA procedures for the chemical constituents of interest.

7) Strict adherence to the established sample collection procedure and preservation protocol is essential to produce results that can be compared with other results from the same well or other wells in a given study.

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ASSESSMENT OF LINER MATERIALS FOR MUNICIPAL SOLID WASTE LANDFILLS

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ABSTRACT

The results of the effects of exposure of 12 lining materials for 56 months to municipal solid waste (MSW) leachate under conditions designed to simulate those that exist at the bottom of a MSW landfill are presented and discussed. The principal liner materials in this study were:

Four admix materials:
- Paving asphalt concrete
- Hydraulic asphalt concrete
- Soil asphalt
- Soil cement

Two asphaltic membranes:
- Bituminous seal
- Emulsified asphalt on a nonwoven fabric

Six flexible polymeric membranes based upon the following polymers:
- Butyl rubber
- Chlorinated polyethylene (CPE)
- Chlorosulfonated polyethylene (CSPE)
- Ethylene propylene rubber (EPDM)
- Low-density polyethylene (LDPE)
- Polyvinyl chloride (PVC)

In addition to the 12 primary liner specimens, 42 samples of membrane materials were also exposed to the leachate as coiled strips buried in the sand above the primary liner specimens.

The 56 months of exposure to the leachate did not increase the permeability of any of the liners. However, the exposure resulted in losses in the compressive strength of the admix liner materials and in softening of the asphaltic materials. It also resulted in swelling of most of the polymeric membranes and in losses in the physical properties of some of the membranes. Several seams lost strength, but the heat-sealed seams, as a group, retained their strength best.

Among the flexible membranes, the low-density polyethylene, a partially crystalline polymer, sustained the least change during the exposure period; however, the low-density polyethylene membrane was thin and susceptible to easy puncturing and tearing which could cause problems during installation and probably cause problems during service under most conditions. The thermoplastic membranes, i.e., chlorinated polyethylene and chlorosulfonated polyethylene, tended to swell most. The vulcanized rubbery liner materials, i.e., butyl and EPDM, changed little during the exposure period, but had the lowest initial seam strength.

This paper also presents the results of immersing specimens of 28 different polymeric membrane liner materials in MSW landfill leachate for 31 months. These 28 liners include variations in polymer, compound, thickness, fabric reinforcement, and manufacturer. The polymeric materials include butyl rubber, chlorinated polyethylene, chlorosulfonated polyethylene, elasticized polyolefin, ethylene propylene rubber, neoprene, polybutylene, polyester elastomer, low-density polyethylene, plasticized polyvinyl chloride, and polyvinyl chloride plus pitch.

INTRODUCTION

This engineering research project was undertaken in 1973 with the primary purpose of assessing the effects of exposure to leachate from municipal solid wastes upon a variety of liner materials. At the time the project was initiated, the lining of landfills with low-permeability materials appeared to be a feasible means of preventing the leachate generated in a landfill from migrating into and polluting surface and groundwaters. Considerable data existed in 1973 as to the service lives of lining materials for water containment and conveyance, but there was concern as to the long-term durability and effectiveness of various liner materials in contact with MSW leachate.

Interim results on this project have been reported as they were available (Haxo and White, 1974; Haxo, 1976; Haxo and White, 1976; Haxo, 1979; Haxo et al., 1979). The effects of one year of exposure were comparatively mild (Haxo and White, 1976) and the data were inadequate to form a basis for making long-range projections of service lives. Consequently, the exposure periods were extended from the original two-year period to 56 months and the scope of the project was expanded to include studies on additional materials and an investigation of simpler methods of assessing the effectiveness and durability of the various liner materials.

This paper reports on the investigations of polymeric membrane liner materials and particularly on the effects of the total immersion of these materials for 31 months in leachate produced in the same simulated landfills used in this project. Water absorption is also discussed. This paper summarizes the findings with respect to exposure testing for the entire project.

METHODOLOGY AND EXPERIMENTAL DETAILS

The basic approach followed in this investigation was to expose a variety of materials to MSW leachate under conditions which simulate those that would be encountered by a liner in real service. The effects of the exposure were determined by measuring appropriate properties of the respective liners as a function of exposure time. All lining materials, except compacted soils and clays, were considered; the latter were studied in other projects.

Two sets of 12 primary specimens were placed in exposure in 24 simulators and retrieved and tested at two exposure times. The principal liner materials that were selected for the exposure study included a wide range of materials that were used at the time for water containment and conveyance. They consisted of admix materials, 2 spray-on asphalt membranes, and 6 commercial polymeric membranes that were available in 1973.

The conditions at the bottom of a landfill that must be simulated for the generation of leachate and for the exposure testing of liner materials include the following features:

- Municipal solid waste.
- A porous soil cover on the liner on which waste can be dumped without damaging the liner and through which leachate generated in the fill can drain.
- The liner, a material of low permeability, which can vary considerably in thickness.
- Compacted sand on which the liner has been placed. The soil has been graded to allow drainage of the leachate.
- Uncompacted native soil.
- Groundwater level, usually well below the landfill.

The design of the simulated landfill and leachate exposure apparatus is illustrated in Figures 1 and 2.

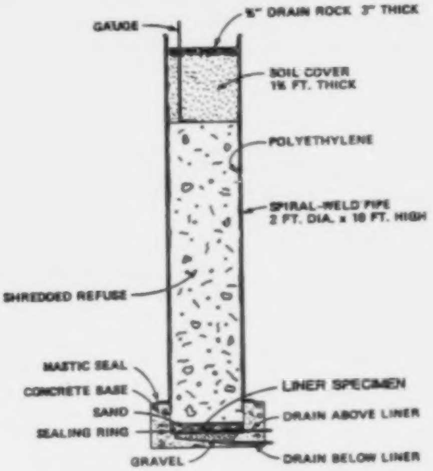
¾" DRAIN ROCK 3" THICK
GAUGE
SOIL COVER
1½ FT. THICK
POLYETHYLENE
SPIRAL-WELD PIPE
2 FT. DIA. x 10 FT. HIGH
SHREDDED REFUSE
MASTIC SEAL
CONCRETE BASE
SAND
SEALING RING
GRAVEL
LINER SPECIMEN
DRAIN ABOVE LINER
DRAIN BELOW LINER

Figure 1. Landfill simulator used to evaluate liner materials exposed to sanitary landfill leachate.

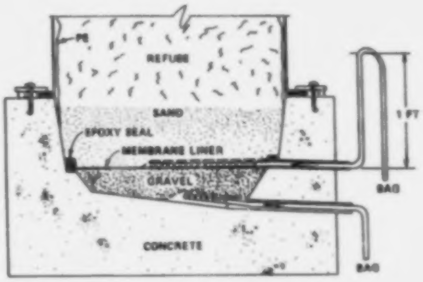
PE
REFUSE
SAND
EPOXY SEAL
MEMBRANE LINER
GRAVEL
1 FT
BAG
CONCRETE
BAG

Figure 2. Base of the landfill simulator in which the liner materials were exposed. The refuse at the bottom of the column was anaerobic. The leachate was maintained at a head of one foot by U-tubes. Plastic bags were sealed at both outlets. Strip specimens of membrane liners were buried in the sand for exposure to leachate.

Approximately 930 pounds of shredded municipal refuse were compacted in each of the 24 simulators. The composition of the refuse has been discussed and described in detail (Haxo and White, 1974; Haxo and White, 1976).

The simulators were brought to field capacity over a period of a month by the addition of tap water. Water equivalent to two inches of rainfall per month was added biweekly to produce the leachate which was used throughout these studies. The average compositon of the leachate at the end of the first year, when the first set of simulators was disassembled and the liner specimens were retrieved and tested, is shown in Table 1.

TABLE 1. ANALYSIS OF LEACHATE*

Test	Value
Total solids, %	3.31
Volatile solids	1.95
Nonvolatile solids	1.36
Chemical oxygen demand (COD) g/l	45.9
pH	5.05
Total volatile acids (TVA), g/l	24.33
Organic acids, g/l	
Propionic	2.87
Isobutyric	0.81
Butyric	6.93

*At the end of the first year of operation when the first set of liner specimens were recovered and tested.

The strength of the leachate decreased with time as is shown in Figures 3 and 4. The compositions of the individual leachates from the 12 simulators, which were retained in operation since the first group of liners were recovered and tested in November 1975, were initially quite uniform. However, in later months the average strength of the leachates decreased significantly and differences in compositions developed. Nutrient (urea) was added with the water to several simulators to increase the strength of the leachate, particularly of the volatile acids; however, the concentration of the acids continued their downward trend (Figure 4).

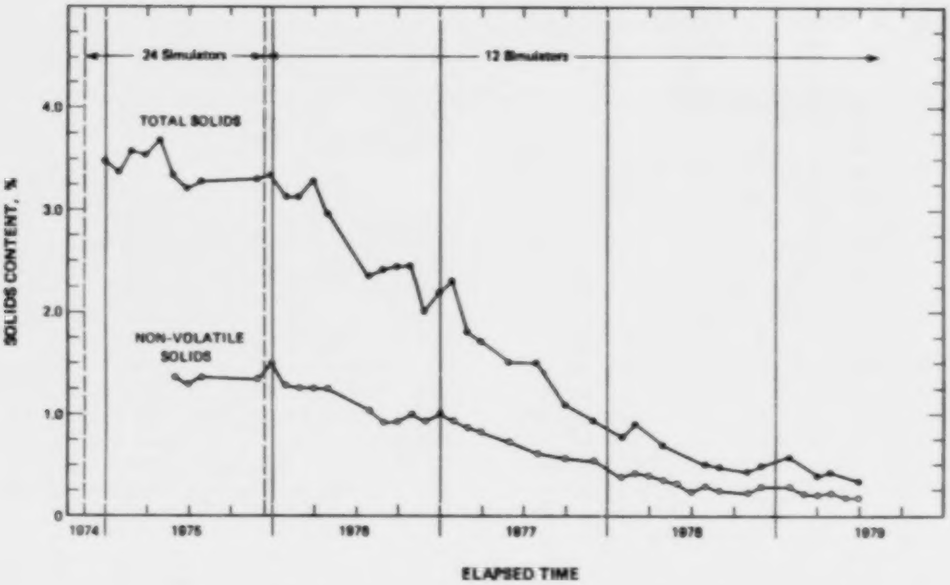

Figure 3. Average solids contents of the leachate produced in the simulators, November 1974 - August 1979. The data for November 1974 - November 1975 are the averages for the leachate from 24 simulators. Twelve simulators were disassembled in November 1975 and, consequently, the data for December 1975 - July 1979 are the averages for the leachates from the 12 remaining simulators.

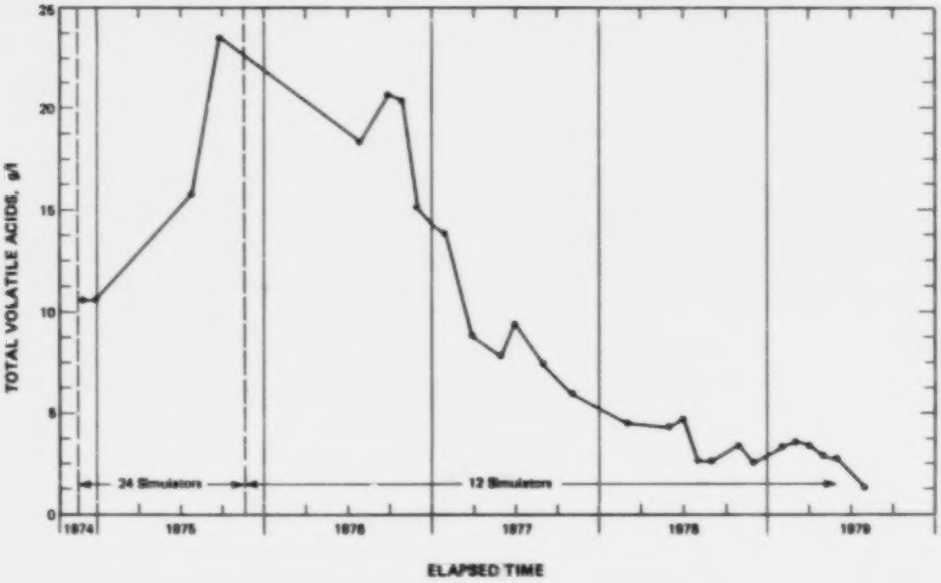

Figure 4. Average total volatile acids content (TVA), as acetic acid, of the leachate produced in the simulators, November 1974 - August 1979. The data for November 1974 - November 1975 are the averages for the leachates from 24 simulators. The data for December 1975 - July 1979 are the averages for the leachates from 12 simulators.

RESULTS OF 56 MONTHS OF EXPOSURE OF LINERS TO MSW LEACHATE

Retrieval of Liners from Simulators

The first set of 12 simulators were dismantled at the end of one year of operation and the samples of the liner materials were recovered and tested (Haxo and White, 1976; Haxo, 1977). The columns of waste of two simulators were lifted off the bases after 43 months of operation and the liner specimens, both asphalt membranes, in the bases were inspected. Also, the strip specimens buried in the sand above the primary liner specimens mounted in the bases of the two simulators were recovered and tested. One of the asphalt membranes, the emulsified asphalt, was sampled and the hole repaired with a cement grout. New strip specimens were buried in the sand and the simulators were returned to operation. All 12 of the simulators in the second set were dismantled after 56 months of operation and the liner specimens, both the primary and the buried, were recovered and tested. The epoxy seals in several of the bases had deteriorated allowing leachate to bypass the liners and enter the bases. In these cases, the permeability of the liners could not be assessed. As most of the epoxy seals withstood the leachate, it is believed that for those cast rings which failed, the mix ratios of the two-part resin were not correct. Before removing or cutting the exposed liner specimens, they were leak-tested by flooding the surfaces and slightly pressurizing the spaces below and any air bubbles were noted. All specimens were kept moist until tested.

Admix Liner Materials

Test results on the four admix liner materials are presented in Tables 2 and 3.

Hydraulic Asphalt Concrete

The hydraulic asphalt concrete specimens were disks 22 inches in diameter and 2 4 inches in thickness. The concrete was hot mixed and compacted in forms outside the bases. The specimens were sealed into the bases with an epoxy resin (Haxo and White, 1974; Haxo and White, 1976). The concrete contained 9% asphalt of 60-70 penetration grade. The aggregate was Watsonville granite proportioned to meet 0.25 inch maximum gradation for dense-graded asphalt concrete. The original voids ratio of the concrete was 2.9%.

At the time the simulators were opened, sand above the liner had hardened in an area essentially coinciding with the area of the glass fabric which covered the drain tube. A dark material had been deposited along both sides of the drain tube, which had sunk into the liner. The entire epoxy ring was light-colored and appeared to be in good condition. No leaks were found when the liner and ring were flooded and pressure applied from below. Six two-inch diameter cores were cut. The bottoms of the cores and the gravel underneath appeared dry.

The unconfined compressive strength of the hydraulic asphalt exposed for 56 months was lower than that exposed for 12 months; the strength at 56 months was only 6% of the strength of an unexposed sample and 13% at 12 months (Table 2). There was no measurable flow through a core of the exposed liner during a four-day run in the back-pressure permeameter, so the coefficient of permeability has been reported as less than 10^{-9} $cm \cdot s^{-1}$. Asphalt extracted from the top of the liner was harder than the original extracted ashpalt, and only slightly harder than that extracted from the liner exposed for only one year. There is some question regarding the high viscosity of the asphalt that was extracted from the bottom part of the liner. This value is being checked. Absorption of water by the asphalt binder affects the compressive strength values of the concrete which are determined without drying of the specimens. Water absorption does not influence the viscosity of the asphalt as the absorbed water is removed during the extraction and recovery process.

Paving Asphalt Concrete

The paving asphalt concrete specimens were prepared in a similar fashion to the hydraulic asphalt concrete specimens. This concrete was hot mixed and contained 7% asphalt (60-70 penetration grade). The aggregate was also Watsonville granite proportioned to meet 0.25 inch maximum

TABLE 2. PROPERTIES OF ADMIX LINER MATERIALS

	Paving asphalt concrete	Hydraulic asphalt concrete	Soil cement	Soil asphalt
Properties after 56 months of exposure to MSW leachate:				
Thickness, inches	2.2	2.4	4.5	4.0
Density, g/cm^3 (lb/ft^3)	2.417 (150.9)	2.385 (148.9)	...	2.019 (126.0)
Voids ratio, vol. voids/vol. solids, %	5.3	3.6	...	...
Water content, g water/100 g dry wt.	1.06	0.96	...	6.51
Compressive strength, psi	258	172	1182[a]	26
Strength retained of original, %	9[b]	6[b]	62[c]	2[b]
Leachate collected below liner during monitoring period:				
Total cumulation, kg	...	3.24[d]	none	12.1
Week leakage began	20th	...	...	1st
Coefficient of permeability[e], cm/sec:				
Initial value	1.2×10^{-8}	3.3×10^{-9}	1.5×10^{-6}[f]	1.7×10^{-3}
After one year exposure to landfill leachate[g] (individual determinations)	7.4×10^{-7} 9.3×10^{-9} 3.0×10^{-9}	$<3.0 \times 10^{-10}$ 3.5×10^{-9}	1.5×10^{-8}[h] 4.0×10^{-7}[i]	1.3×10^{-8} 2.8×10^{-8}
After 56 months exposure to landfill leachate[g] (individual determinations)	$<10^{-9}$	$<10^{-9}$	1.2×10^{-5} 2.7×10^{-5}[j] 4.3×10^{-7}[k] 1.2×10^{-5}[l]	8.6×10^{-8} 2.7×10^{-7}

[a] Tested exposed specimen contained 10% cement.

[b] Original specimen molded from unexposed sample stored 12 months.

[c] Original specimen tested on unexposed molded specimen 12 months after molding, specimen contained 12% cement.

[d] On dismantling of simulator, leak was found to be in epoxy seal.

[e] Back pressure permeameter tests made on cores cut from compacted liner specimen.

[f] Molded test specimen. Cores satisfactory for testing could not be cut from the compacted liner specimen.

[g] Value for core specimens cut from liner.

[h] Top section of Core 3.

[i] Bottom section of Core 4.

[j] Top and bottom sections of Core 1, which included a leak detected before coring.

[k] Top and bottom sections of Core 4 where leaks were detected before coring.

[l] Top and bottom sections of Core 3.

TABLE 3. CHANGES IN PROPERTIES OF ASPHALT IN ADMIX MATERIALS AND MEMBRANES DURING 12 AND 56 MONTHS EXPOSURES TO MSW LANDFILL LEACHATE

	Asphalt concrete			Hydraulic asphalt concrete			Soil asphalt			Bituminous seal		Asphalt on non-woven fabric	
	Whole core	Top	Bottom	Whole core	Top	Bottom	Whole core	Top	Bottom				
Type of asphalt	60-70 pen paving grade asphalt			60-70 pen paving grade asphalt			Slow curing liquid asphalt SC-800 grade			Catalytically blown asphalt		Emulsified asphalt	
Viscosity at 25°C, sliding plate viscometer, MP:													
At shear rate of 0.05 sec^{-1}:													
Original	5.1[a]	...	...	2.1[a]	...	...	0.02[b]	...	...	8.5	...	4.5[b]	...
After 12 months	8.8	...	...	3.3	...	...	0.04	...	...	10.04	...	2.9	...
After 56 months	...	9.1	10.7	...	4.2	13.5[c]	...	0.005	0.012	...	17.4	...	3.1
Change	+3.7	-4.0	+5.6	-1.2	-2.1	+11.4	+0.02	-0.015	-0.008	+1.9	+8.9	-1.6	1.4
Change, %	+73	+78	+110	+57	+100	+548	+100	-75	-40	+22	+104	-36	-31
Penetration at 25°C:													
Original	44[d]	...	...	62[d]	...	...	538[b,e]	...	...	36	...	46[b,e]	...
After 12 months[e]	36	...	...	52	...	...	390	...	...	34	...	55	...
After 56 months[e]	...	35	33	...	47	30	...	1020	668	...	27	0.53	...
Change	-8	-9	-11	-10	-15	-32	-148	+482	+352	-2	-9	+9	+7
Voids ratio (volume voids/volume solids x 100):													
Original	6.4	...	...	2.9	...	...	10.4	...	...	...	...	...	...
After 12 months	4.2	...	...	1.9	...	...	26.1	...	...	...	...	...	...
Change	-2.1	...	...	-1.0	...	...	+15.7	...	...	...	...	...	...
Softening point, °C :													
Original	...	...	...	...	...	...	...	...	...	89	...	...	...
After 12 months	...	...	...	...	...	...	...	...	...	89	...	...	...
After 56 months	...	...	...	...	...	...	...	...	...	101	...	...	...

[a] Calculated from penetration data.
[b] Asphalt extracted from unexposed specimens stored 12 months.
[c] Data questionable and being checked.
[d] Asphalt before mixing was 68 penetration, i.e., ca. 1.7 MP.
[e] Calculated from viscosity data.

gradation for dense-graded asphalt. The specimens were disks 2.2 inches thick and 22 inches in diameter. They were compacted hot into forms outside the bases and sealed into the simulator bases with epoxy adhesive (Haxo and White, 1974; Haxo and White, 1976).

On opening the simulator above the liner, it was found that much of the sand had hardened with a dark cementing agent. Some of the hardened sand had loose sand under it. About two-thirds of the epoxy ring was in very poor condition; swollen, flaking, soft and very dark; however, no leak was found when the liner and ring were flooded and pressure applied below. Six two-inch diameter cores were cut. The bottoms of the cores and the gravel underneath appeared dry. The permeability of the asphalt concrete decreased during the exposure period. The viscosity of the asphalt increased significantly.

Compressive strength of the asphalt concrete liner after 56 months of exposure was only 9% of the strength of the unexposed concrete of the same composition. After 12 months of exposure, the liner had retained 15% of the strength of the unexposed liner, thus indicating continued deterioration of the asphalt concrete. There was no measurable flow through a core of the exposed liner during a three-day run in the back-pressure permeameter. Since this time interval would be adequate for determining a coefficient of permeability of 10^{-9} $cm \cdot s^{-1}$, we report less than 10^{-9} $cm \cdot s^{-1}$ for this liner. The properties of the asphalt extracted from it were very similar to those measured after 12 months of exposure, indicating that very little change to the asphalt occurred during the additional exposure. Asphalt extracted from a concrete specimen of the same original batch after 56 months of exposure to air and weather had a viscosity of 37.0 MP at 25°C and a calculated penetration of 21. Thus the exposure in the simulator was much less severe on the asphalt per se than normal weather, which probably reflects the anaerobic environment at the bottom of a landfill.

Soil Asphalt

The soil asphalt specimens were four inches thick; and were compacted directly in the bases. The soil asphalt consisted of 100 parts soil mixed with 7.0 parts of liquid asphalt, Grade SC-800 (Haxo and White 1974; Haxo and White, 1976). A ring seal of epoxy resin was placed at the top of the compacted liner to prevent possible bypassing of the liner by the leachate at the edge of the specimen.

When the simulators were dismantled, sand was scraped aside just enough to examine the condition of the liner, which was very soft and had the appearance of stiff mud. Black, soft concretions were around much of the rim and near the drain tube. Also, spots of concretions were scattered over the surface of the liner. The fabric had been cut just past the end of the drain tubes, and the liner was indented, apparently by the rim of the simulator during removal. The epoxy ring appeared to be in good condition, though part appeared to contain too much gravel.

A black deposit along the drain tube was sticky and tarry. In the pressure leak test, leaks were found in the liner and along the epoxy rings. All of which were within three to four inches of the rim. The leak areas were mostly from areas of oily concretions on the surface. The liner material lost almost all of its compressive strength during the exposure. At the time of compaction, the unconfined compressive strength measured 1218 psi, at the end of a year it measured 15 psi, and at 56 months it measured 26 psi.

A core cut from the center was dry. Samples of gravel from the bottom of the core holes appeared wet.

When a core, which came from an area where leaks were detected, was tested in the back-pressure permeameter, it was deformed by the confining pressure used in the permeameter. The value determined for coefficient of permeability, approximately 10^{-7} $cm \cdot s^{-1}$, may not be reliable because of deformation of the specimen.

Asphalt extracted from the bottom half of a core was slightly less viscous than the original SC-800 grade liquid asphalt, and asphalt extracted from the top half of the core was much lower in viscosity than the original. This effect, and the presence of a small amount of oily material in the sand covering the liner, leads to the conclusion that the waste in the simulator must have contained a low-viscosity, nonvolatile, oily substance which contaminated and softened the liner.

Soil Cement

The soil cement specimens were compacted directly in the bases and sealed with a ring of epoxy resin. These specimens were 4.5 inches in thickness and were made of 95 parts of soil, 5 parts of Kaolinite clay, 10 parts Portland cement, and 8.5 parts water which were thoroughly mixed before compaction (Haxo and White, 1974; Haxo and White, 1976).

The sand layer was very shallow because of the thickness of the liner. When opened, it was found that only a small amount of sand was hardened, mostly along a line extending perpendicular from the middle of the drain tube to the rim. There was a dark deposit along the drain tube, and most of the slots were plugged. The epoxy ring appeared to be in good condition. The liner was hard, and similar to portland cement concrete. One small leak, about one inch from the rim, appeared in the liner when it was flooded and pressure applied from below. Six two-inch diameter cores were cut. Core 1 was cut at the location of the above leak, and included the leak. All cores from this liner were recovered in two or three pieces, and many showed horizontal voids, possibly indicating boundaries between lifts. Core 1, close to the rim, cut into the lower rim and broke off, though there was some soil cement below that point. The hole from the first core did not drain. Water from the other five cores holes ran out the bottom drain. The bottoms of all the cores and the gravel underneath all appeared wet, though that may have been due to the water used for coring.

Compressive strength of the soil cement liner after 56 months of exposure to leachate was the same as for the liner specimens recovered and tested after one year of exposure. Permeability tests of the top and bottom sections of Core 1, which included a leak detected before coring, gave K values of 1.2×10^{-5} and 2.7×10^{-5} $cm \cdot s^{-1}$, approximately an order of magnitude more permeable than the molded specimen tested originally. For Core 3, a similar value was determined for the bottom section, but the top was less permeable, 4.3×10^{-7} $cm \cdot s^{-1}$. The latter value is in the same range as the values determined for cores of the liner tested after one year (Haxo and White, 1977).

Asphaltic Membranes

Results on the exposure tests of asphaltic membranes are presented in Tables 3 and 4.

Bituminous Seal

The bituminous seal membrane was cast in place at 425°F on a sand bed covering the aggregate in the base of the simulator It was a catalytically blown canal lining asphalt and was approximately 0.30 inch in thickness (Haxo and White, 1974; Haxo and White, 1976). A ring of epoxy resin was formed around the top edge of the liner.

On examination at the time the refuse and sand were removed, the canal lining asphalt membrane had some areas in which the liner was weak, "cheesy", and cracked on bending; other areas were still tough and flexible. When the liner was examined in the laboratory, after 36 days of storage in a sealed polyethylene bag, the differences were less pronounced. Two weak areas were identified, one near the rim and one at the free end of the drain tube. The weak areas were not as tender as when examined originally. Samples from the each of the above weak areas and from "normal" area were removed for determination of volatile loss and viscosity. Volatile loss of the sample from the weakest area was about three times that of the other two areas. Viscosity of the asphalt from all three areas was approximately equivalent. Since the sample was heated in the preparation procedure for viscosity determination, any water and volatile organic materials were driven off. Reversible effects caused by the presence of water or other volatiles, therefore, are not detected in the viscosity test.

The asphalt membrane liner tested after 12 months of exposure had not changed in softening point and had increased in viscosity only slightly from the original asphalt. After 56 months of exposure, both softening point and viscosity had increased, indicating that the asphalt had hardened during exposure, so far as its properties in the dry state were concerned. The viscosity increase of

TABLE 4. ASPHALT MEMBRANE LINERS AFTER 56 MONTHS EXPOSURE TO MSW LANDFILL LEACHATE

	Bituminous Seal Catalytical-blown Asphalt				
	Weak (near drain tube)	Normal	Weak (near rim)	Avg	Fabric and asphalt emulsion
Volatiles, %					
47 days at 23°C	1.49	1.26	3.71		7.32
2 hours at 110°C	1.42	1.46	4.55		7.95
Softening point, °C (°F)		101 (214)			
Tests on extracted asphalt:					
Viscosity at 25°C, MP					
At 0.05 s^{-1}	18.5	17.8	15.8	17.4	3.1
At 0.001 s^{-1}	162	161	152	158	3.4
Shear susceptibility	0.56	0.55	0.58	0.56	0.03
Penetration at 25°C (calculated from viscosity)	27	27	28	27	53

the liner exposed to leachate was less than for a sample removed from the upper surface of the same lot of asphalt remaining in the fiberboard carton stored alongside the simulators since they were constructed, showing that exposure to air caused a greater increase in viscosity than did exposure to leachate. While the measurements made in the dry state indicate good retention of properties, the very tender, "short" and "cheesy" condition observed when the liner was first removed causes some concern for the ability of the liner to withstand minor earth movements when it is in the saturated condition existing beneath a landfill.

Asphalt Emulsion on Nonwoven Fabric

The emulsified asphalt on nonwoven fabric was an asbestos filled anionic asphaltic emulsion on a nonwoven polypropylene fabric. This membrane was prepared by the supplier in sheet form. Disk specimens 22 inches in diameter, were mounted in the bases and sealed with epoxy resin. The liner was 0.30 inch thick.

The liner of non woven polypropylene fabric coated with asphalt emulsion had absorbed nearly 8% water (or other volatiles) during 56 months of exposure to leachate vs. almost 5% for the liner tested after one year exposure. Viscosity of the extracted asphalt was lower than the original, but differed very little from the viscosity after one year.

A patch of a nonshrinking cement grout, applied to fill the hole left where a sample was cut from the liner at 43 months, was in good condition, well sealed to the edges of the hole, and appeared to be functioning satisfactorily. No seepage of leachate was collected below the liner during the 13 months that the patch was in place.

Polymeric Membranes

Primary Test Specimens

The six polymeric membranes that were exposed as primary liners in the bases of the simulators were:

Butyl rubber
Chlorinated polyethylene (CPE)
Chlorosulfonated polyethylene (CSPE)
Ethylene propylene rubber (EPDM)
Low density polyethylene (LDPE)
Polyvinyl chloride (PVC)

In view of the importance of seams in field performance of a membrane liner, a seam was incorporated across the middle of each specimen. The changes in the physical properties and seam strengths of the six primary polymeric membrane specimens after 56 months of exposure to MSW leachate are presented in Table 5. The retention of original values of selected properties are presented in Table 6.

Overall, the changes in the physical properties of the membranes resulting from 56 months of exposure were relatively minor. All of the membranes softened to varying degrees during the first 12 months, as shown by hardness data in Table 6, probably due to the swelling by the leachate. In the interval of time to 56 months, the PVC, CSPE, and CPE membranes rehardened slightly, possibly indicating, in the case of the PVC membrane, loss of plasticizer and, in the case of the CSPE and CPE membranes, crosslinking of the polymers. They all recovered most of their tensile properties that were lost due to the initial softening. These materials were all thermoplastic and unvulcanized.

Of the six polymeric membranes, the LDPE film best maintained original properties during the exposure period (Table 6). It also absorbed the least amount of leachate. However, this membrane, which was 10 mils in thickness, probably has too low a puncture resistance for use in lining a landfill. This deficiency was also observed in the handling of the sheeting which was used to line the steel pipes in which the ground refuse was compacted.

The butyl and EPDM liners changed somewhat more in physical properties than did the LDPE during the exposure period. In particular, they maintained their stress-strain properties. Some softening did occur in the 44 months interval between the removal of the first set of liner specimens at 12 months and the second set at 56 months.

The fact that, except in the bases in which the epoxy resin sealing ring deteriorated, no leachate appeared below the membrane liners during the exposure period indicates a very low permeability of the polymeric membranes.

The strength of the seams of the PVC, the CSPE, and the CPE membrane liner specimens sustained substantial losses, with the losses greater for the CSPE and CPE membranes. The seam shear strength of the butyl and EPDM liners initially increased, but then decreased in strength. The LDPE retained best its original seam strength throughout the period; it was heat-sealed and the test values were low (the failure was just outside of the seam, thus indicating potential weakness at that point).

Much better seam strength values were obtained with the buried strip test specimens, as discussed in the next section.

Strip Membrane Specimens Buried in the Sand above Primary Specimens

In addition to the six primary membrane specimens, 42 smaller specimens of membrane liners and other polymeric compositions were buried in the sand above the primary specimens and exposed to leachate. These additional specimens allowed us to make comparisons between materials from different sources, different constructions and different thicknesses, as well as exposure to one side and both sides of the test specimens. During most of the exposure period, a head of at least a foot of leachate was maintained on the surface of the primary liner; consequently, these specimen were essentially immersed in the leachate with both sides of the strips in direct contact with leachate. The buried specimens were in the form of spirally coiled strips, 2 1/8" x 20". At one end of each a lap adhesive joint approximately 2" x 2" was incorporated which could be tested in peel and in shear. Thus, various adhesive bonding systems could be tested.

The buried specimens included the following compositions and variations:

1. Samples of four of the six primary liner materials which were mounted in the bases of the simulators with the same adhesive systems plus additional adhesive systems.
2. Additional liner materials of the same six polymers, but varying in source, thickness, and fabric reinforcement.
3. Membranes of additional polymers which are potentially useful as liners, i.e. neoprene, polybutylene, and polypropylene.

TABLE 5. PROPERTIES OF POLYMERIC MEMBRANES AFTER 56 MONTHS OF EXPOSURE TO LEACHATE GENERATED IN SIMULATED MSW LANDFILL

Polymer		Butyl	CPE	CSPE	EPDM	LDPE	PVC
Matrecon number		7	12	6R[a]	18	21	17
Nominal thickness, mils		63	31	34	51	11	20
Type of compound[b]		VZ	TP	TP	VZ	TP	TP
Thread count, ends per inch (epi)		...	...	8 x 8	...	...	...
	Direction of test						
Analyses:							
Volatiles		2.37	7.61	13.9	5.74	1.95	2.08
Extractables (after volatiles removed)		9.76	5.09	3.35	28.27	3.37	34.39
Solvent		MEK	nC_7H_{16}	DMK	MEK	MEK	CCl_4+ CH_3OH
Physical properties:							
Tensile strength of fabric, ppi	Machine	...	...	37	...	...	...
	Transverse	...	...	35	...	...	...
Tensile strength of polymer, psi	Machine	1490	2270	1890	1520	2610	2920
	Transverse	1440	1650	2330	1360	2560	2560
Elongation at break, %	Machine	420	360	220	380	470	340
	Transverse	385	410	245	370	610	335
Set at break, %	Machine	12	160	50	5	350	55
	Transverse	11	160	65	7	470	68
S-100, psi	Machine	320	1060	800	350	1390	1330
	Transverse	350	510	845	305	1050	1130
S-200, psi	Machine	720	1460	1670	860	1500	1940
	Transverse	780	820	1980	735	1150	1680
S-300, psi	Machine	1160	1910	...	1240	1760	2620
	Transverse	1210	1220	...	1200	1240	2330
Tear strength, Die c, ppi	Machine	185	180	185	135	450	300
	Transverse	185	160	220	145	360	270
Hardness:							
Duro A, instant reading		57	75	74	54	81	72
10 sec. reading		51	70	70	51	81	70
Duro D, instant reading		...	...	...	...	34	...
10 sec. reading		...	...	...	...	26	...
Puncture strength:							
Thickness of specimen, mils		64	37	37	49	9.8	22
Stress, lb		50.0	51.8	58.2	41.5	17.1	31.3
Elongation, in.		1.25	0.98	0.86	1.19	1.24	0.84
Seam strength:							
Seam system		Cement 8800 (lab)	Solvent, 50 THF: 50 Toluene (lab)	Cement 280Z (lab)	Solvent 374+ Cement MAG (factory)	Heat seal (lab)	Solvent 6079+ L-1552 (factory)
Shear, ppi		17	10	17	18	11	22
Locus of failure[c]		AD-LS	AD-LS	AD	AD	SE	AD-LS
Peel:							
Average, ppi		3.4	2.9	1.8	7.1	12	5.6
Maximum, ppi		3.8	4.5	3.2	7.6	12	7.5
Locus of failure[c]		AD	AD	AD-LS	AD	SE	AD-LS

[a] Fabric reinforced.

[b] VZ = Vulcanized; TP = Thermoplastic.

[c] Locus of failure code:
AD = Failure within the adhesive.
AD-LS = Failure between adhesive and liner surface.
SE = Failure at seam edge.

TABLE 6. RETENTION OF PHYSICAL PROPERTIES AFTER 56 MONTHS OF EXPOSURE TO LEACHATE GENERATED IN SIMULATED MSW LANDFILL

Polymer		Butyl	CPE	CSPE	EPDM	LDPE	PVC
Matrecon number		7	12	6R	18	21	17
Nominal thickness, mils		63	31	34	51	11	20
Type of compound		VZ	TP	TP	VZ	TP	TP
Thread count, ends per inch (epi)		...	...	8 x 8	...	...	...
	Direction of test						
Tensile strength of fabric, % ret.	Machine	...	...	99	...	...	...
	Transverse	...	...	100	...	...	...
Tensile strength of polymer, % ret.	Machine	103	92	107	101	154	111
	Transverse	99	79	145	94	99	102
Elongation at break, % ret.	Machine	86	120	92	90	147	126
	Transverse	112	70	82	78	70	88
Set after break, % ret.	Machine	125	80	64	38	198	81
	Transverse	164	70	82	78	70	88
S-100, % ret.	Machine	91	87	81	100	109	106
	Transverse	121	98	94	87	102	100
S-200, % ret.	Machine	94	80	97	113	102	93
	Transverse	128	98	137	97	111	90
S-300, % ret.	Machine	94	78	...	111	105	...
	Transverse	121	102	...	93	111	...
Tear strength, % ret.	Machine	103	67	58	75	108	85
	Transverse	103	67	77	80	100	85
Puncture strength:							
Stress, % ret.		112	110	169	105	128	121
Elongation, % ret.		102	94	151	83	163	122
Hardness, Duro D:							
Duro A, pt. diff. inst. reading		-2	-10	-5	-3	NT[c]	-9
10 second reading		0	-7	-5	-3	NT[c]	-6

[a] R = Fabric reinforced.

[b] Type of compound = Vulcanized (VZ) or Thermoplastic (TP).

[c] NT = no test.

4. A series of five pieces of gasket sheeting of different rubber compositions and molded slabs of two thermoplastic rubbers.

5. At the time that two simulators were lifted from the base to inspect the conditions of the asphaltic membrane liner specimens, 10 strip specimens of newly-obtained membranes were placed in the sand above the asphalt liners and were exposed for the last 13 months of the test.

Selected results of the tests of the buried strip specimens are presented in Tables 7 and 8.

The changes in physical properties of four of the buried specimens are compared with those of the same materials as the primary specimens in Table 7. Inasmuch as the buried specimens contacted the leachate on both sides, the effects of the exposure to leachate could be expected to be greater than that encountered by the primary specimens which contacted the leachate from only one side. The results given in Table 7 show that, in three of the four buried specimens, the absorption of leachate was greater than that of the primary specimens. In the fourth case, butyl, the absorption was essentially the same.

With respect to the effects upon physical properties, the results varied among the four materials. Overall, however, the buried samples tended to soften more than did the primary samples. The major differences between the two exposures were in seam strengths of some of the membranes. This probably reflects the differences in the preparation of the test specimens. For example, the adhesion of the buried CSPE sample was significantly better than that of the primary sample; on the other hand, the reverse was true of the EPDM specimens.

Although there is variation in the results, the data indicate that specimens immersed in a landfill can indicate the effects of leachate and can be used as a measurement of the effects of leachate on liners.

There are variations, however, among the liners based upon a given type of polymer, as can be seen by inspection of the data for PVC membranes (Table 8). As can be seen from the data, the PVC membranes vary considerably in initial composition and properties. For example, the plasticizer contents of five PVC membranes, as measured by extractables, varied from 32.3 to 38.9%. In tensile strength, the variation among 9 different PVC membranes was from 2435 to 3400 psi and elongation ranged from 200 to 385%. The retention values also varied considerably among the different PVC membranes, indicating a difference possibly in the migration of the plasticizer out of the compounds. These results indicate the variation among the different PVC membranes and show that recognition of this should be given in their selection for specific applications.

Of particular interest, are variations in the leachate absorption by different liners of the same type polymer. In the case of the PVC membranes, the absorption ranged from 0.37% to 6.75%. In the case of the CSPE membranes, swelling ranged from 8.7% to 21%. EPDM varies from 5.1% to 9% in leachate absorption.

IMMERSION STUDY OF POLYMERIC MEMBRANES

The comparable results of the first year of exposure tests of the buried strips of the membrane materials with those of the primary liners, indicated that the complete immersion of a specimen in leachate would be a feasible method of exposing liners for test purposes (Haxo, 1979).

The availability of leachate generated in the simulated MSW landfills made it possible to use, for immersion testing, the same fluid that was used in the exposure testing in the simulators of the primary and buried specimens. This gave us the opportunity to develop correlation among three types of exposure conditions.

Consequently, a study was undertaken to evaluate membrane liner materials only, as admix materials do not lend themselves to tests of this type. The basic design of the immersion system was discussed in detail previously (Haxo, 1977) and is shown in Figure 5. Three sets of 28 specimens of each membrane were immersed in the leachate so that they could be removed in sets from the system after three exposure intervals. For this study, the specimens were removed and

TABLE 7. COMPARISON OF PROPERTIES OF PRIMARY AND BURIED SPECIMENS AFTER 56 MONTHS OF EXPOSURE TO LEACHATE IN THE MSW LANDFILL SIMULATORS

Polymer	Butyl		CPE		CSPE		EPDM	
Matrecon number	7		12		6R[a]		18	
Nominal thickness, mils	63		31		34		51	
Type of compound[b]	VZ		TP		TP		VZ	
Thread count, ends per inch (epi)	...		...		8 x 8		...	
Type of specimen	Primary	Buried	Primary	Buried	Primary	Buried	Primary	Buried
Analyses:								
Volatiles (2 h at 105°C), %	2.37	2.02	7.61	10.1	13.9	17.0	5.74	6.47
Extractables (after volatiles removed), %	9.76	9.79	5.09	5.73	3.35	3.84	28.3	30.0
Solvent[c]	MEK		nC_7H_{16}		DMK		MEK	
Physical properties[d]:								
Tensile strength of fabric, ppi	...	...	...	...	37	41	...	...
Tensile strength of polymer, psi	1490	1510	2270	2205	1980	2620	1520	1505
Elongation at break, %	420	435	360	295	220	200	380	495
Set at break, %	12	16	160	165	50	53	5	15
S-100, psi	320	300	1060	1165	800	1245	350	295
S-200, psi	720	680	1460	1665	1670	2620	860	655
S-300, psi	1160	1090	1910	2185	...	...	1240	975
Tear strength, Die C, ppi	185	185	180	215	185	205	135	175
Hardness:								
Duro A, instant reading	57	65	75	71	74	70	54	55
10 sec. reading	51	53	70	64	70	67	51	47
Puncture strength:								
Thickness of specimen, mils	64	66	37	38	37	40	49	58
Stress, lb	50.0	38.0	51.8	49.3	58.2	57.4	41.5	41.5
Elongation, in.	1.25	1.14	0.98	0.97	0.86	0.84	1.19	1.42
Seam strength:								
Bonding system	Cement 8800 (lab)		Solvent, 50 THF: 50 Toluene (lab)		Cement 280Z (lab)		Solvent 374+ Cement MAG (factory)	
Shear, ppi:	17	39.5	17	na	10	6.92	18	14.9
Locus of failure	AD-LS	AD	AD	na	AD-LS	SE	AD	AD
Peel:								
Average, ppi	3.4	3.0	2.9	2.0	1.8	16.0	7.1	0.4
Maximum, ppi	3.8	4.2	4.5	2.6	3.2	22.8	7.6	0.7
Locus of failure[c]	AD	AD-LS	AD	AD	AD-LS	AD	AD	AD

[a] Fabric reinforced.

[b] VZ = Vulcanized; TP = Thermoplastic.

[c] Solvent: MEK = Methylethyl ketone; DMK = Dimethyl ketone = Acetone; nC_7H_{16} = n-heptane.

[d] Tensile properties all in machine direction.

[e] Locus of failure code:
AD = Failure within the adhesive.
AD-LS = Failure between adhesive and liner surface.
SE = Failure at seam edge.

TABLE 8. PROPERTIES OF PVC MEMBRANE SPECIMENS BEFORE AND AFTER EXPOSURE TO LEACHATE IN MSW LANDFILL SIMULATORS

Matrecon number	10	11	15	17	19-1	67	88	92A	93
Simulator number		1			1	11	12	12	11
Nominal thickness, mils	34	31	11	20	21	21	20	22	11
Exposure time, months	56	56	43	43	56	13	13	13	13
Analytical properties:									
Volatiles, %									
Initial	0.30	0.15	0.47	0.08	0.05	0.03	0.17	0.23	0.12
Final	6.32	1.97	1.92	2.10	0.51	0.66	0.67	1.29	1.73
Change	+6.02	+1.82	+1.45	+2.02	+0.46	+0.63	+0.50	1.06	1.61
Extractables, %									
Initial	...	33.90	...	37.25	38.91	...	33.50	...	32.26
Final	30.53	31.37	33.33	34.39	36.13	36.25	30.64	30.72	29.34
Change	...	2.53	...	2.86	2.78	...	2.85	...	...
Physical properties:									
Thickness, mils									
Initial	32	30	10	20.5	22	22	20	20	11
Final	34	31	11	20	24	21	20	21	11
Change	+2	+1	+1	-0.5	+2	-1	0	+1	0
Tensile, psi									
Initial	3400	3140	2360	2640	2780	3020	3395	2435	3035
Final	3140	2955	2180	2760	2765	2995	3590	2655	3145
Retention, %	92.4	94.1	92.4	104.5	99.5	99.2	105.7	109.0	103.6
Elongation, %									
Initial	280	325	200	270	330	385	325	245	350
Final	330	315	280	380	340	345	310	335	315
Retention, %	117.9	96.9	140.0	140.7	103.0	89.6	95.4	136.7	90.0
S-200, psi									
Initial	2610	2295	2360	2080	1840	1820	2610	2110	2055
Final	2420	2355	2000	1800	2020	2150	2730	1895	2410
Retention, %	92.7	102.6	89.7	86.5	109.8	118.1	104.6	87.3	117.3
Tear resistance (Die C)									
Initial	390	380	290	352	295	340	463	435	362
Final	450	425	380	385	345	360	560	420	390
Retention, %	115.4	111.8	131.0	109.4	116.9	105.9	121.0	96.0	107.7
Hardness, Duro A, 10 sec. rdg.									
Initial	82	80	72	76	72	75	80	82	78
Final	85	85	69	72	75	73	78	75	75
Retention, %	+3	+5	-3	-4	+3	-2	-2	-7	-3
Puncture strength (20 ipm)									
Stress, lb									
Initial	...	39.5	12.8	25.8	...	27.8	28.6	27.4	15.9
Final	61.2	59.7	15.4	29.6	35.9	35.1	34.0	33.4	18.9
Retention, %	...	151.1	120.3	114.7	...	122.7	118.9	121.9	118.9
Elongation, in.									
Initial	0.62	0.62	0.61	0.69	...	0.68	0.56	0.62	0.55
Final	0.78	0.80	0.70	0.88	0.80	0.78	0.68	0.74	0.56
Retention, %	126	129	115	128	...	115	121	119	102

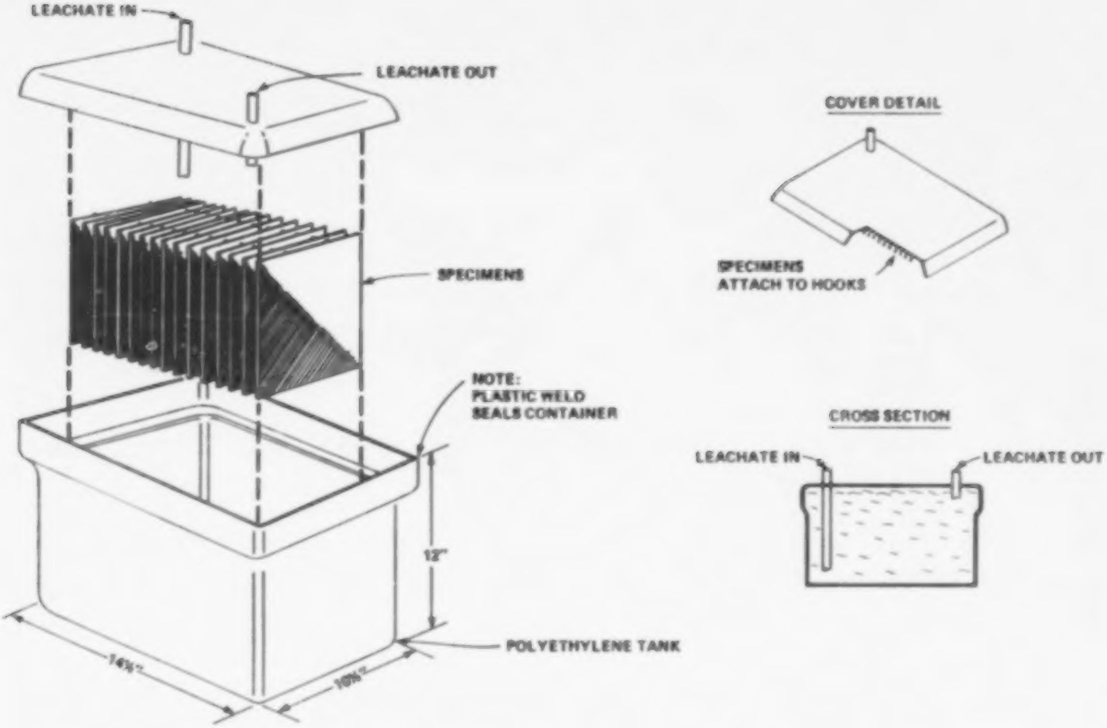
LEACHATE IN
LEACHATE OUT
SPECIMENS
NOTE: PLASTIC WELD SEALS CONTAINER
12"
POLYETHYLENE TANK
14¼"
10½"
COVER DETAIL
SPECIMENS ATTACH TO HOOKS
CROSS SECTION
LEACHATE IN
LEACHATE OUT

Figure 5. Individual polyethylene immersion tank, showing method of holding specimens and the tank and outlet for the leachate.

tested at intervals of eight, nineteen, and thirty-one months. The leachate from the 12 remaining simulators was collected and blended biweekly and slowly pumped through the cells holding the exposure specimens.

The tests performed on the lining materials before exposure and at three subsequent intervals were:

- Weight
- Dimensions
- Tensile strength, in machine and transverse directions, ASTM D412.
- Hardness, ASTM D2240.
- Tear strength in machine and transverse directions, ASTM D624, Die C.
- Puncture resistance, FTM 101B, Method 2065.
- Volatiles at 105°C, ASTM D297.

Selected results are summarized in Tables 9 and 10.

The buried and primary specimens generally swelled equally in the first 12 months that they were exposed in the simulated landfills. In many cases, effects of eight months of immersion were equal to the 12 months of exposure in the simulated landfills. On the basis of the initial data, it appears that the immersion in the leachate is quite comparable to the exposure in the simulated sanitary landfills (Haxo, 1979).

In Table 9, the data on swelling of the liners during the immersion in leachate are arranged by polymer type. The range of values for a given polymer is indicated for eight, nineteen, and thirty-one months. The data show that the

TABLE 9. SUMMARY OF THE EFFECTS OF IMMERSION OF POLYMERIC MEMBRANE LINERS IN LEACHATE FOR 8, 19, and 31 MONTHS

Liner material	Number of liners in tests	Leachate absorption, %			Tensile strength, % original			Elongation, % original			Change in hardness, Duro A, points		
		8 mo.	19 mo.	31 mo.	8 mo.	19 mo.	31 mo.	8 mo.	19 mo.	31 mo.	8 mo.	19 mo.	31 mo.
Butyl rubber	1	1-2	3.5	25	90-97	89-94	92	104-106	99	90-92	0	0	-1 to +2
Chlorinated polyethylene	3	8-20	26-28	25-28	80-115	81-106	78-106	64-135	76-108	71-103	-5 to 1	-8 to -2	-11 to -1
Chlorosulfonated polyethylene	3	13-19	16-27	19-32	82-124	95-132	103-138	97-107	77-94	69-86	-20 to -4	-26 to -5	-21 to -3
Elasticized polyolefin	1	0.1	2.0	8	86-94	96-107	98-106	91-92	102	96-98	0	-2	-1
Ethylene propylene rubber	5	1-13	1-12	8-24	90-01	86-93	94-113	76-138	83-146	88-138	-1 to +2	-2 to -1	-3 to +5
Neoprene	4	1-19	3-32	5-88	69-100[a]	60-102	68-105	82-103[a,b]	76-104	78-146	-11 to +5	-12 to +3	-18 to +4
Polybutylene	1	0.1	0.8	-7	96-99	94-95	84-97	96-97	96	86-89	-3	-5	-3
Polyester elastomer	1	2.0	1.9	16	99-115	52-94	81-90	101-108	92-94	80-96	-4	-6	-3
Low density polyethylene	1	0.6	1	3	110-180	92-149	118-161	96-181	67-192	100-168	-7	-10	-9 to -8
Polyvinyl chloride	7	1-3	1-6	4-24	91-110	91-111	87-117	98-139	100-129	79-120	-2 to +1	-6 to +1	-6 to +3
Polyvinyl chloride + pitch	1	6	8	14	92	88-93	101-104	109-133	94-117	80-103	-2	+1	+1 to +7

[a] Data on fabric reinforced neoprene liner number 42 were not included.
[b] Unexposed material broke at 150% elongation.

TABLE 10. RETENTION OF MODULUS[a] OF POLYMERIC MEMBRANE LINER MATERIALS ON IMMERSION IN LANDFILL LEACHATE

Polymer	Liner no.	S-200, psi		Retention of original value, %		
		Unexposed	31 mo.	8 mo.	19 mo.	31 mo.
Butyl rubber	44	685	670	86	90	98
Chlorinated polyethylene	12	1330	1260	85	89	95
	38	1205	1250	89	90	104
	86	810	1080	98	106	133
Chlorosulfonated polyethylene	3	745	420	54	46	57
	6[b]	1550	...	114	134	195
	85	1770	2300	77	108	130
Elasticized polyolefin	36	970	1100	104	109	113
Ethylene propylene rubber	8	690	790	126	124	114
	18	760	880	110	109	116
	83[b]	845	880	107	107	104
	91	855	840	91	92	98
	41	1040	1090	100	99	105
Neoprene	9	1235	940	79	77	76
	37	1635	1620	100	100	99
	42[b]	...	...	...	...	...
	90	1190	1370	105	114	115
Polybutylene	98	3120	3310	101	101	106
Polyester elastomer	75	2735	2735	102	98	100
Polyethylene	21	1260	1340	106	102	106
Polyvinyl chloride	11	2120	2080	87	85	98
	17	1965	1850	80	84	94
	19	1740	1950	89	94	112
	40	1720	1790	91	91	104
	67	1700	1990	92	105	117
	88	2400	2420	79	88	101
	89	2455	2580	96	95	105
Polyvinyl chloride + pitch[c]	52	1020	...	85	86	...

[a] Average of stress at 200% elongation (S-200) measured in machine and transverse directions.

[b] Fabric reinforced.

[c] S-100 values given; original and subsequent exposed specimens failed at less than 200% elongation.

swelling varies from polymer to polymer. The table also shows the variation of the retention of tensile strength and elongation, and changes in membrane hardness. Also, within a given polymer type there is variation due to compound differences. CPE, CSPE, and neoprene showed the greatest variations within a polymer type. On the other hand, the seven PVC liner specimens varied only a few percent. In all cases, except the polybutylene, the swelling values for 31 months of immersion were higher than the previous values.

The effect on the modulus (stress at 200% elongation) of the same materials is shown in Table 10. The effects appear to be related to the degree of swell which the specimens have undergone. Those specimens which swell little vary little in tensile strength and modulus. Again, the PVC membranes have a low spread in values and retained their original tensile and modulus. Overall, the polyolefins, such as polyethylene, polybutylene, and elasticized polyolefin exhibit the lowest swelling and highest retention values.

The effect of immersion in leachate up to 31 months appears to have a relatively mild effect upon most of the membranes as shown for tensile strength and retention versus immersion time in Figure 6.

The changes in properties, e.g. tensile strength, appear to be generally related to the amount of swell which the membranes have undergone during the immersion. This is illustrated in Figure 7, in which retentions of tensile strength have been plotted against the leachate absorption for the liner materials.

WATER ABSORPTION STUDY

Since MSW leachate is primarily water which, in itself, can affect the properties of polymeric materials particularly on prolonged exposure, water absorption studies were conducted using ASTM Test Method D570 (ASTM, 1975). Absorption tests were run at room temperature and at 70°C. The objective of these studies was to determine the correlation of swelling by leachate with that by water. Also, water is a standard reference fluid for swelling tests. In addition, it was desired to determine whether the swelling of the liner materials would plateau, that is, come to equilibrium on extended immersion times. Results of water absorption of 11 membrane liners on immersion in water up to 186 weeks, both at room temperature and at 70°C are presented in Table 11. The order of the results for the two temperatures were generally the same; however, one of the major differences was with one of the PVC liners (No. 11), which yielded the lowest swell at room temperature, but was sixth at 70°C. The liner materials that had the lowest swell were the polyvinyl chloride, the elasticized polyolefin, and the ethylene propylene rubber; those that swelled the most were the membranes based on neoprene, CSPE, and CPE. The CPE and the EPDM essentially reached equilibrium swell at 70°C in the 100 weeks. A PVC liner appeared to have reached equilibrium swell; however, it hardened because of loss of plasticizer. It appears that several of the other liner materials, particularly the neoprene and CSPE, will continue to swell for considerable lengths of time.

DISCUSSION

Asphalt-Containing Lining Materials

The 56 months of exposure of lining materials to MSW leachate indicate that there may be problems in the use of asphaltic materials for the lining of municipal solid waste landfills. In this study four forms of asphalt were included:

- 60-70 Penetration-grade paving asphalt used in hydraulic and pavement concretes
- A catalytically-blown asphalt used in a membrane
- A slow-curing liquid asphalt used in soil asphalt
- An emulsified asphalt used in an asphalt membrane

The permeability of the various asphaltic liner materials did not increase and the changes in properties of the extracted asphalts were small compared with with the changes that occurred in air aging of the same asphalt; nevertheless, all five liner materials containing asphalt changed significantly in the properties of the liners themselves, i.e:

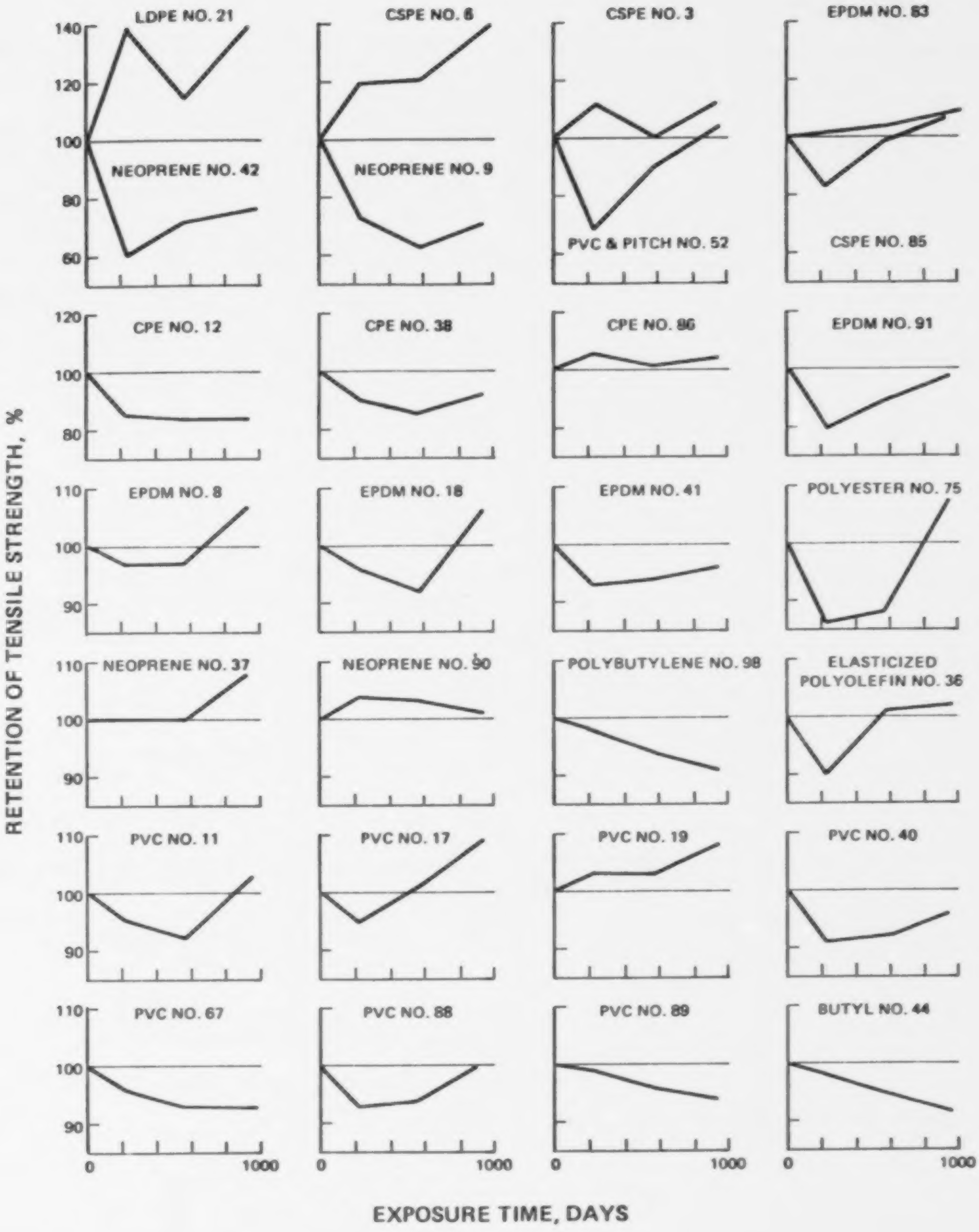

Figure 6. Retention of tensile strength of the individual polymeric membranes as a function of immersion time in landfill leachate. Tensile strength values based upon the average data obtained in the machine and transverse direction. Liner numbers and initial tensile strength for each liner are shown. Data are given for 8, 19, and 31 months.

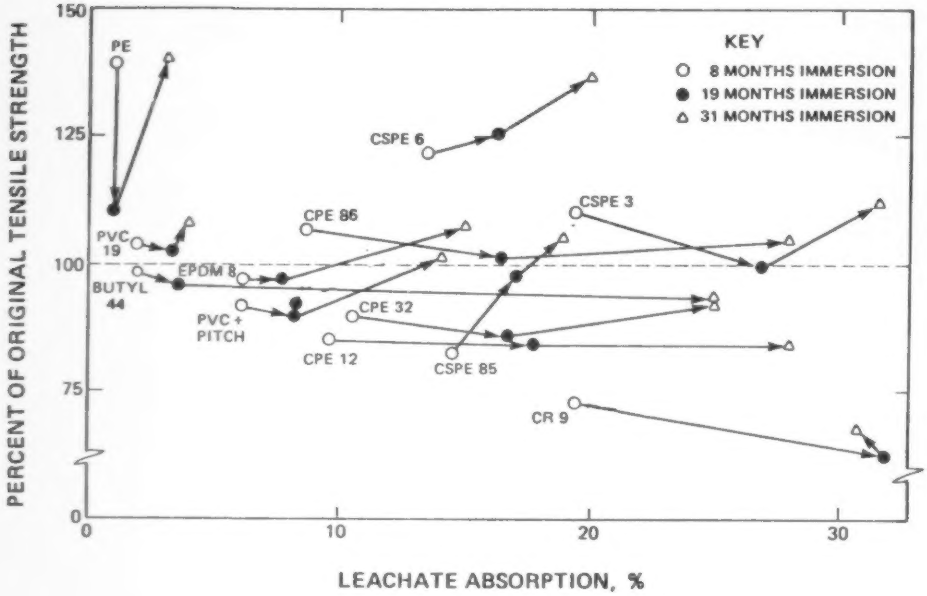

Figure 7. Retention of average tensile strength of membrane liner materials during immersion in landfill leachate as a function of swelling by the leachate.

1. There was major absorption of leachate or water which resulted in significant decreases in the compressive strength of the concretes and the soil asphalt.
2. The canal-lining asphalt, based upon catalytically-blown asphalt, absorbed water or leachate and became cheesey and lost elongation.
3. The emulsified asphalt absorbed more water or leachate than did the catalytically-blown asphalt, but it appeared to retain its strength because of the presence of the fabric. The greater absorption may be due to the presence of the asbestos and water soluble salts.

In this study, the exposure period was 56 months; however, indications are that these asphaltic-based liners would continue to change with prolonged exposure. The complete loss of strength and elongation can be expected to have adverse effects upon the functioning of lining materials, particularly if any ground movement takes place.

There are no ready explanations for the deterioration of all of the asphaltic lining materials. Possibly, a high salt concentration in the asphalt may cause swelling by osmosis. The effect on concrete may reflect the particular aggregate that is used. although the Watsonville granite has not been found to cause "stripping", which is a problem encountered with many aggregates when used in road building. The continuous and prolonged exposure of these asphaltic materials to water and to leachate is a condition which is not normally encountered in many asphaltic products. Even in the case of water reservoir linings, the exposure is often intermittent. Also, some asphalt may be more resistant to exposure to MSW leachate, but specifications would have to be developed to be certain that the correct types are used.

Soil Cement Liners

The exposure of soil cement to the MSW leachate did not appear to have any adverse effects upon the soil cement liner specimens. Actually, the soil

TABLE 11. WATER ABSORPTION OF SELECTED MEMBRANE LINER MATERIALS AT ROOM TEMPERATURE AND AT 70°C[a]

		Water absorbed, % by weight											
		At room temperature						At 70°C					
Polymer	Liner number	1 week	11 weeks	44 weeks	100 weeks	159 weeks	186 weeks	1 week	11 weeks	44 weeks	100 weeks	159 weeks	186 weeks
Butyl rubber	57	0.82	3.22	4.50	6.4	10.92	10.76	4.62	17.54	53.9	103.2	...	141.3
Chlorinated polyethylene	77	1.63	5.53	10.2	12.5	16.85	16.51	15.9	58.4	140.0	179.3	178.89	117.80
Chlorosulfonated polyethylene	6	3.44	6.97	10.9	16.3	20.81	19.42	22.1	131.0	245.6	370.5	438.16	471.40
Elasticized polyolefin	36	0.39	0.52	0.0	4.5	0.93	0.38	0.36	0.45	0.57	8.7	-4.19	-8.48
Ethylene propylene rubber	8	0.50	1.30	1.56	2.25	3.48	35.3	1.11	3.55	10.8	17.8	12.59	18.53
	26	1.20	1.84	1.49	3.85	3.61	3.61	1.44	4.52	11.20	17.4	21.08	22.60
Neoprene	43	3.80	13.62	37.8	75.1	67.20	63.11	14.1	107.0	240.0	(b)	(c)	...
	82	2.43	8.29	18.5	32.1	4.40	52.48	8.11	47.4	191.4	295.0	349.95	(d)
Polyester elastomer	75	1.07	1.05	0.67	1.31	1.61	0.36	1.28	1.10	0.72	0.22	0.79	(d)
Polyvinyl chloride	11	1.29	1.10	0.70	1.25	1.74	-0.21	5.59	12.13	39.2	87.4	126.31	164.28
	59	1.59	2.34	2.43	2.98	4.53	1.55	4.87	8.25	24.0	25.5[e]	26.17	26.86

[a] ASTM D570-63 specimens 1 x 2 inches in deionized water.
[b] Specimens began to disintegrate between 44th and 69th weeks.
[c] Specimens have become hard, indicating loss of plasticizer.
[d] 151 weeks of exposure.
[e] Specimen disintegrated.

cement probably gained in strength and decreased in permeability during the exposure. The soil cement liner, after preparation and cure, could not be cored and it was necessary to use our laboratory molded specimens for determining the properties of unexposed soil cement. These specimen probably gave high results. The exposed liner specimens, after 12 and 56 months, yielded good cores which could be tested for permeability and strength. It must be recognized that soil cement is brittle and the specimens that were exposed in this project were only 22 inches in diameter. Large areas of soil cement, such as used as bases for highway pavement, crack every few feet which would eliminate the use of soil cement for the lining of large water disposal sites, which can not tolerate major cracking of a lining.

Polymeric Membrane Lining Materials

Butyl Rubber

The butyl rubber sheeting was vulcanized and unreinforced and had a nominal thickness of 63 mils. During the 56 months of exposure, it absorbed somewhat over 2.7% by weight of leachate and did not appear to change in thickness. The physical properties of the sheeting changed little when the data in the machine and transverse directions are averaged. The differences between the machine and transverse directions are not consistent. Generally, the properties in the machine direction changed downwards and, in the transverse direction, increased.

During the exposure, puncture resistance increased. The seam strength, however, was poor at the start, but did not change during the course of the exposure. The bonding method was a low-temperature vulcanizing cement.

Chlorinated polyethylene (CPE)

The CPE liner was thermoplastic and not crosslinked, had an initial thickness of 31 mils, and was unreinforced. During the 56 months of exposure it absorbed 6.7% by weight of leachate, but the absorption had levelled off. The properties generally decreased with the exposure, although the values are not large. However, of the six membranes which were tested, this membrane showed the greatest change with exposure time. Also, seam strength exhibited a substantial decrease in both peel and shear.

Chlorosulfonated polyethylene (CSPE)

The CSPE sheeting also was thermoplastic, was fabric reinforced, and had an initial thickness of 32-36 mils. During the exposure, it had the greatest absorption of leachate and when tested at the end of the exposure period had a volatiles content of almost 14%. As with the other liners, the absorption appeared to have levelled off. The tensile strength of the fabric, which was reported to be nylon, did not change during the exposure period. The base compound, however, increased in tensile strength, puncture resistance, and modulus. On the other hand, the tear strength decreased. The strength of the seams, which were made with a bodied cement, decreased substantially during the exposure. This results was not consistent with that obtained on the buried strips which yielded much higher values.

Ethylene Propylene Rubber (EPDM)

The sheeting of EPDM was based on a vulcanized compound. It was unsupported and had a nominal thickness of 51 mils.

After 56 months of exposure, the sheeting had a volatiles content of 5.75%, which is equivalent to an absorption of 6.1%. The extractables, after removal of the volatiles, fell from 31.8% to 28.3%. This quantity of extractables is a compounding oil, a portion of which was dissolved in the leachate. The effect of the exposure upon physical properties was minor. There was essentially no change in tensile strength, puncture resistance,and only modest loss in elongation and tear. None of these losses was near 50%. The bond strength was low at the beginning and appeared to have improved during the exposure period.

Low-density polyethylene (LDPE)

The LDPE sheeting had a thickness of 11 mils. It was used to line the simulators as well as being used as a test specimen in the base of two of the simulators.

This material had a low volatiles content but, during exposure, it appeared to have absorbed some leachate. With respect to the physical properties, there was almost no change during the exposure period. There was a slight increase in puncture resistance which was comparatively low due to its low thickness.

Polyvinyl chloride (PVC)

The polyvinyl chloride sheeting that was tested as a primary specimen had a thickness of 21 mils and was unsupported. Commercial sheeting ranges from 10 to 30 mils.

During 12 months of exposure, this sheeting appeared to absorb 3.55% by weight of leachate and, after 56 months, the absorption had dropped to 2.08%. At the same time, the PVC lost plasticizer. The effect of the exposure on this sheeting was minor. The tensile strength and elongation remained unchanged; however, the S-200 stress decreased at 12 months and increased at 56 months, possibly reflecting the difference in absorbed leachate. This effect was also noted in hardness and tear strength. The strength of the seam, which was factory prepared using a proprietary cement, was low, but improved slightly during the exposure period, from 4 to 5.6 pounds per inch in the peel mode.

Changing Materials and Technology

The polymeric membranes that were selected when the project began were all commercially available at the time. However, several of these are no longer available, either because the companies which regularly supplied the materials are no longer manufacturing the sheeting or have changed their formulations and construction. Also, the seaming techniques and materials are undergoing changes. Consequently, the data presented must be used with discretion when applied to present materials.

ACKNOWLEDGEMENTS

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FIELD VERIFICATION OF LINERS

Assessment of Long-term Exposed Liner Materials From Municipal Solid Waste Landfills

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ABSTRACT

The objective of this partially completed project is to study the physical and chemical condition of clay, asphalt and PVC liners subjected to continued long-term exposure of leachate and decomposing refuse. This paper discusses the procedures followed in selecting the initial test sites, the development of sampling methodology to obtain samples from liner materials buried under refuse fills and from the soils beneath the liner, and the institutional and legal difficulties encountered. Physical and chemical testing of collected polymeric samples is still in progress by EMCON Associates and its subcontractor, Matrecon, Inc.

INTRODUCTION

Of prime importance in disposing of municipal solid wastes is the protection of ground water and surface water from the environmental impact of leachate--liquid that has percolated through waste and has extracted dissolved or suspended materials from it, thereby becoming contaminated. Using impervious lining materials to intercept and control leachate offers a promising means of reducing or eliminating water degradation from this source, and is recommended by the EPA for consideration as a control mechanism.

Traditionally, naturally occurring clay materials have been used to control leachate from disposal facilities with an abundant supply of native impermeable material. However, frequently it was found that the native material was not as impervious as had been assumed, there were areas of higher permeability than anticipated, the production of leachate was higher than anticipated, the quality of leachate was worse than predicted, or the site was in a more sensitive ground-water system than the investigations had determined. Furthermore, available future sites may not be as impervious as desired. It is therefore imperative that alternative containment materials be available to permit disposal of waste products and assure containment of leachate.

Over the past 30 years we have seen the development of many chemically formulated products which are, for all practical purposes, impermeable to liquid and gaseous flows. Among these are: polyethylene (PE), polyvinyl chloride (PVC), chlorinated polyethylene (CPE), chlorosulfonated polyethylene (CSPE), butyl rubber, and various asphalt mixtures.

Chemical and hazardous waste disposal and storage facilities have been subject to increasing review and public scrutiny. Recently they have also been brought under severe, but properly deserved, regulatory review similar to that afforded sanitary landfill facilities.

The issues associated with the selection and use of lining materials include, but are not limited to the following:

- compatibility of liner material with the waste products and environment.
- aging characteristics of the liner
- construction techniques
- economics

A performance issue with liners is the possibility of degradation after prolonged exposure to leachate. Leachate may chemically or physically attack liner materials, thus impairing their integrity. Unfortunately there presently exists no standard method of testing to predict the life of various lining materials. There is strong circumstantial evidence that liner life can exceed 20 years; research to date on liner materials exposed to landfill leachate has indicated only minimal changes in the properties of liner materials. Clearly, however, more information is needed.

The primary objective of this project is to obtain lining sample specimens from existing landfills to determine the changes in the properties of the lining material as a function of age and to validate data being developed through laboratory research. Specifically, the project consists of two phases, with four subtasks each. Phase I will be a study of landfill liners at disposal sites having clay, asphalt, or PVC liners.

Originally four sites were selected by EPA for Phase I of the study. These sites were selected from approximately 30 candidate sites identified by another EPA contractor. Preliminary approval had been secured from the site owners by EPA for use of these sites in the research project; however, the contract provided that other candidate sites could be considered.

If the results of Phase I demonstrate significant changes in liner characteristics (original over aged), then Phase II may be initiated. In Phase II, up to six additional sites will be selected for sampling and testing. The subtasks are: (1) obtain final permission from site owners to conduct sampling, (2) obtain liner samples and repair the lining material after extraction, (3) test and evaluate the samples, and (4) compare original liner material with the extracted samples and produce a final report.

Knowledge gained by this study will be valuable to designers, owners and operators of future landfills, as well as to regulatory agencies. It is believed the results will provide valuable assurance that lined refuse disposal facilities can be constructed and operated without inflicting damage to the environment.

STUDY APPROACH

It was recognized at the start of the project that the recovery of liner samples in a sensitive environment would not be an easy task. The liners had been installed initially to prevent degradation of the underground water resources. The sampling techniques therefore included the replacement or repair of the liner so as to continue the protective features.

Prior to proceeding with the task of obtaining final permission to utilize specific sites, a review was made of available information on the four selected sites and other sites throughout the country. The ideal site would be one with complete data and records regarding: (1) the design, installation and performance of the liner material, (2) methods of site operation, (3) types, age and thickness of waste placed over the liner, and (4) occurrence, quantity and character of leachate. Other selection criteria included: information on soils and hydrogeology beneath the liner, variety of types and thickness of liner material, interest and cooperation of regulatory agencies, and attitude of site owner/operator. Telephone contacts were made with the originally selected site owner/operators to obtain more data and to seek their continued interest in the project. During this process one of the owners declined to participate.

Using the above criteria, the review of site information and screening of owner/operator attitudes resulted in our recommending four candidate sites as being most beneficial to the goals and objectives of the study - only one of the originally named four sites remained as a candidate. Later, another site had to be substituted when a municipally-owned site declined to participate just before the site visits were to occur.

Task 1 - Obtain Final Permission

Each candidate site was visited by Mssrs. John G. Pacey (EMCON) and Henry Haxo (Matrecon) to familiarize themselves with the site characteristics and explain proposed liner sampling and repair plans to the owners/operators. Site information and operating plans were obtained, where available; geotechnical and design reports were reviewed; photographs were taken; and persons knowledgeable about site operations were interviewed. Particular emphasis was placed on obtaining specifics about the liner material used; types, age, and thickness of wastes placed over the liner; methods of operation; hydraulic head on the liner; quality and history of leachate in contact with the liner; and the availability of excavating and drilling equipment.

Discussion also entailed identifying regulatory agencies, permit requirements, public relations, release of information and legal and liability constraints. These items are discussed in greater detail elsewhere in this report.

Based on data obtained from these initial site visits, the sampling and repair methodology was developed. In all cases the exact depth to the liner was unknown, which meant that the excavations or drilling operations would have to proceed slowly to explore for the liner without destroying the integrity of the liner material. Telephone conversations were held with drilling contractors in the vicinity of the various candidate sites to discuss the feasibility of the suggested sampling and repair techniques.

After the sampling and repair procedures were developed and approved by the EPA Project Officer, the final details were explained to the site owner and written approval was requested. The responses and conditions encountered at this stage of the work are discussed later.

Task 2 - Obtain Sample and Repair

The intent of this task is to obtain samples of the various in-place liner materials and the natural soils beneath the liner to ascertain any movement of leachate into or across the liner materials. Soil samples are also to be obtained from similar soils outside the influence of the landfill to serve as background data for comparison purposes. Samples are to be of sufficient size to conduct the tests identified in Task 3. Samples of the original lining material and/or specifications are to be obtained where possible.

In addition, grab samples from various depths of the refuse are to be obtained to record the general condition of the refuse. Leachate, if present, will also be sampled.

After the liner sample has been obtained, repairs must be made to the liner to prevent the downward movement of any present or future leachate.

Sampling Methodology - Clay Lining

In sampling a clay liner, the proposed methodology involves the use of a hollow stem auger (Figure 1), boring to a point approximately 5 feet above the anticipated depth of liner. A split spoon sampler will be driven beyond the auger in 18-inch increments, with the auger following every advancement. The sampler will be withdrawn after each incremental advancement to ascertain the refuse/clay interface. The auger will then be advanced to a depth of approximately 1 foot into the liner. Shelby Tubes will be driven into the clay to obtain samples to a depth 10 feet below the refuse. After sampling, bentonite will be placed in the boring as the auger is withdrawn to prevent escape of any leachate above the liner.

Sampling Methodology - Asphalt Lining

The asphalt lining at the selected site is reported to have an 8-inch soil cover. The proposed sampling procedure (Figure 2) uses a hollow stem auger and split spoon sampler, as previously discussed, to identify the depth of soil cover over the liner. Once this is deter-

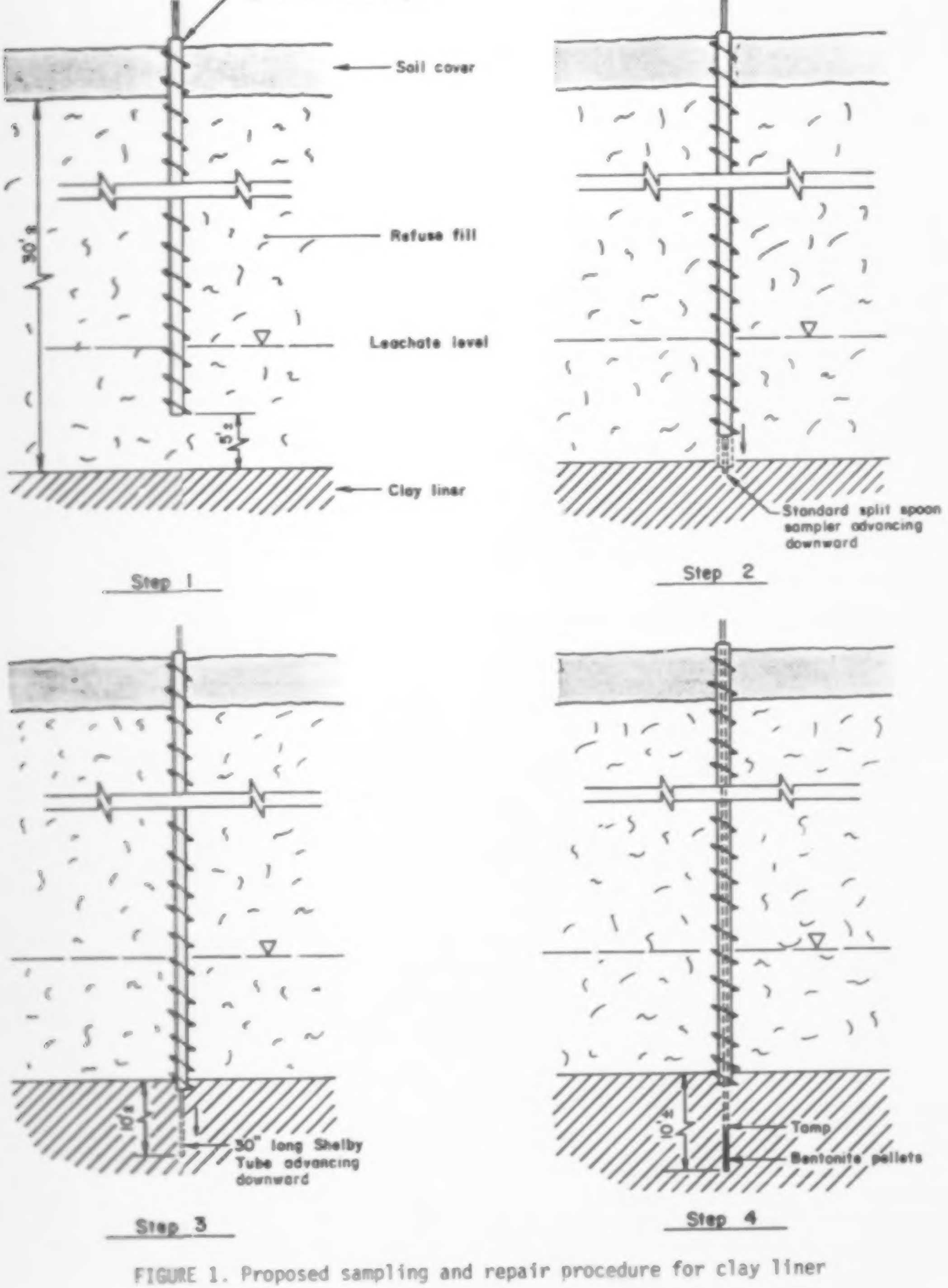
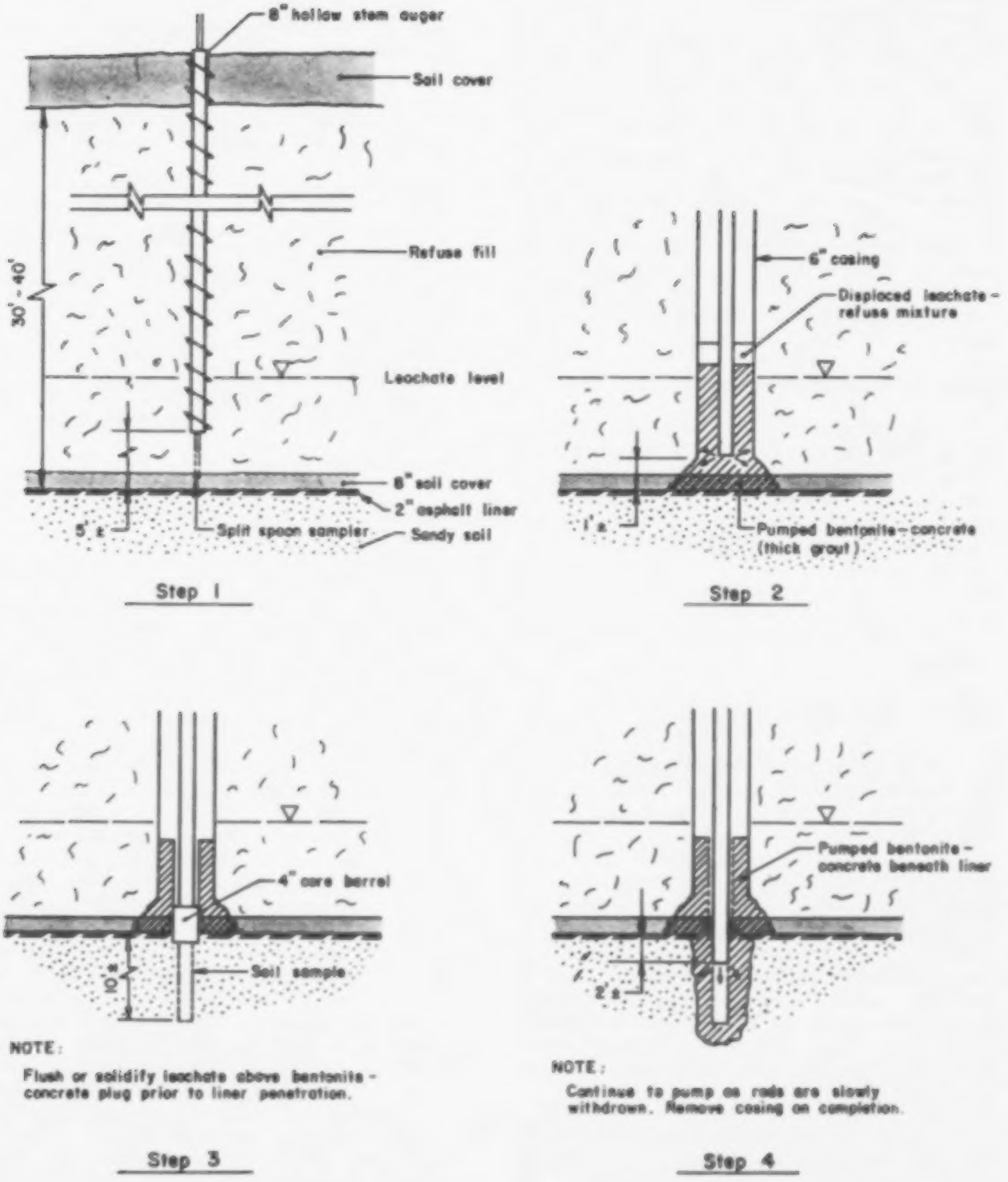

8" hollow stem auger
Soil cover
Refuse fill
Leachate level
Clay liner
30'±
5'±
Step 1
Standard split spoon sampler advancing downward
Step 2
10'±
30" long Shelby Tube advancing downward
Step 3
10'±
Tamp
Bentonite pellets
Step 4

FIGURE 1. Proposed sampling and repair procedure for clay liner


8" hollow stem auger
Soil cover
30'-40'
Refuse fill
Leachate level
8" soil cover
2" asphalt liner
5'±
Split spoon sampler
Sandy soil
6" casing
Displaced leachate - refuse mixture
1'±
Pumped bentonite - concrete (thick grout)
4" core barrel
Soil sample
10'±
Pumped bentonite - concrete beneath liner
2'±

Step 1

Step 2

NOTE:
Flush or solidify leachate above bentonite - concrete plug prior to liner penetration.

Step 3

NOTE:
Continue to pump as rods are slowly withdrawn. Remove casing on completion.

Step 4

FIGURE 2. Proposed sampling and repair procedures for asphalt liners

mined, a 6-inch casing is installed in the hole to within 1 foot of the liner. A bentonite-concrete slurry is injected into the casing to provide sealing against any leachate present. This slurry will be allowed to rise in the casing until its level equals or exceeds the leachate level in the refuse. Any leachate above this bentonite-concrete plug shall be flushed out of the casing or be solidified before proceeding. A 4-inch core barrel is drilled through the plug to retrieve the liner sample. Soil samples will be taken with a split spoon sampler to a depth of 10 feet below the liner. Repairs will be made by pumping bentonite-concrete slurry in the hole up to the top of the previously placed plug. Casing will be withdrawn.

Sampling Methodology - PVC Lining

One of the two sites selected is a relatively shallow fill; sampling was therefore accomplished by excavating through the refuse to the liner with a small backhoe. Shelby Tubes were driven into the subsoil to obtain samples at depths below the liner. The other site was a sludge pond. Excavation at this site was made with a front-end loader. Shelby Tubes were also used for the soil samples.

Task 3 - Testing and Evaluation

This work is to determine any changes in the physical and chemical properties of the lining materials over time. The characteristics and properties of the liner soils will be compared with the characteristics and properties of the background soils to determine the physical and mechanical changes with depth.

Polymeric material testing is to include water permeability, thickness, tensile strength and elongation at break, hardness, tear strength, creep, water absorption or extraction, puncture resistance and density.

Tests on asphaltic materials will include permeability, density and voids, water swell, compressive strength and viscosity.

Chemical tests on soil samples beneath the liners will analyze for pH, Hg, Pb, Zn, Cd, Fe, Cl, COD, Na, NH_4, K and Mg. The intent of this testing is to develop adsorption data. Soils will also be subjected to the following physical tests: permeability, density and voids, water swell, and compressive strength.

Limited chemical analysis will be performed on leachate samples.

Task 4 - Comparison and Reporting

After testing of sample specimens has been completed, and original lining material specifications have been reviewed, the two sets of data will be compared. Where appropriate, comparison with the results of previous and ongoing EMCON/Matrecon studies will be made. The objective is to determine the potential serviceable life of the lining material.

PERFORMANCE OF WORK

In the early discussions with site owners/operators in the Fall of 1979, it became obvious that obtaining final permission would be difficult and time consuming. The ability to meet local and state regulations and the upcoming RCRA "inventory" of sites was uppermost in the minds of each operator. Their sensitiveness to any potential for adverse publicity was understandable. Some owners/operators of both privately and municipally owned sites refused to participate, although they were interested in the results of the study. Others were willing to participate provided the data would be kept confidential and not be available for use against them by the public or regulatory agencies. Since these conditions could not be fully assured, the time allotted for the work effort has been extended and the work redirected to gain limited knowledge of liner performance. To date only two sites have been used to obtain liner and soil samples. Types of polymeric liners sampled are polyethylene (PE), chlorosulfonated polyethylene (CSPE), chlorinated polyethylene (CPE) and polyvinyl chloride (PVC). In addition, a sample from a clay material placed above a polymeric liner is being tested. The materials sampled have been exposed to an adverse environment and, possibly, leachate for up to eight years.

Laboratory testing is underway, and the results should be available early in 1981.

CONCLUSIONS AND RECOMMENDATIONS

Based on a visual examination of the liners exposed during sampling, there were no obvious failures of the polymeric materials, except for some tears apparently made while uncovering the samples. It should be recognized that those areas sampled are small when compared to the total lined area. Since the laboratory work is not yet complete, no conclusions can yet be made regarding physical and chemical changes to the liners.

The ability to sample liner materials is hindered by the same regulations that encouraged the liners to be placed in the landfill. The owner/operator wants legal and financial guarantees that: (1) any adverse information developed remain confidential, (2) his business image not be damaged in any way, (3) he receive full liability protection from sampling and excavation activities and accidents, (4) repairs be made to satisfy his requirements and those of the regulatory agencies, and (5) deficiencies found during the investigation not be used against the site by EPA, local regulators or the public. In summary, the owners/operators believe their sites to be successful and active business ventures, presumably operating in compliance with all existing laws. Participation in this type of project holds the operation up to public scrutiny and places even the best operation in possible jeopardy from weak or incomplete media coverage and unfounded lawsuits. In effect, the advice from their legal counsels is that there is little to gain and a lot to lose by participation in the study.

Added to these concerns are the consultants' liabilities in contracting for work to be performed by local drilling companies performing unique procedures for the first time. There is also the problem and cost of proving whether or not any subsequent contamination of the underground water resources was the fault of the sampling and repair work, the owner's activities, or some extraneous condition or activity. The consultant may also be subjected to the costs of third party legal actions beyond his control.

A team effort is needed if we are to obtain liner samples at landfills which contain leachate. The owners/operators, consultants, and subcontractors and other participants need to be protected from unfounded legal actions and financial risks affecting their livelihood. Perhaps the EPA can find ways to guarantee confidentiality of information, and provide funds to help correct deficiencies discovered and to indemnify participants from legal actions. It is doubtful that any private landfill owner/operator and most public entities will grant permission to sample liners without such incentives.

EFFECT OF SANITARY LANDFILL LEACHATE ON THE ACTIVATED SLUDGE PROCESS

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ABSTRACT

Synthetic sanitary landfill leachate was spiked into a conventionally designed activated sludge wastewater treatment plant and the process control problems evaluated. The synthetic leachate simulated high organic strength, 20,000 mg/l COD, execreted from a young landfill. The monitoring of the activated sludge process included oxygen uptake rate, mix liquor solids, specific oxygen utilization, sludge settling characteristic, and sludge production. The limiting factor in treating leachate was found to be sludge bulking.

INTRODUCTION

It is estimated that the citizens of the United States dispose of over 500 billion pounds of municipal refuse each year. The major method of disposal is by sanitary landfill. It is estimated that in the United States there exists 250,000 acres of active sanitary landfill. There are many problems associated with sanitary landfills, one of which is leachate. As water passes through a landfill, soluble material is extracted from the refuse and forms leachate. The leachate contains organic acids and heavy metals. If the leachate were released directly to the environment, it would be equivalent to dumping 2 billion pounds of BOD per year into the nations streams and rivers. The leachate, if released untreated into the environment, would obviously cause fish kills and pollute the public water supplies.

In 1976 Congress inacted the Resource Conservation and Recovery Act. This placed pressure on sanitary landfills to collect and treat leachate. The EPA-Office of Solid Waste is developing alternative methods of disposing of leachate. One treatment method under investigation is to discharge leachate into a municipal sewer system and to treat the leachate with the municipal sewage at the Municipal Sewage Treatment Plant.

OBJECTIVES

The objectives of this project are to evaluate the impact of sanitary landfill leachate on a conventional activated sludge process and identify cause and effect relationships between the leachate addition and process performance.

CHARACTERISTICS OF SANITARY LANDFILL LEACHATE

The organic constitute of sanitary landfill leachate results from the biological decomposition of organic refuse within a landfill. It has been proposed that the biological decomposition process is composed of four phases. (1) The first phase is characterized by the decomposition of high molecular weight humic carbohydrate material. The second phase is characterized by the biological decomposition of free volatile fatty acids and the production of compounds containing the carbonyl group (C = O) and amino acids. The third phase is characterized by the decomposition of the compounds containing the carbonyl group and amino acids and the production of high molecular weight carbohydrate compounds. The fourth phase is characterized by the slow decomposition of the carbohydrates produced in the third phase. The organic material present in leachates from landfills in the first and second phase are readily treated by biological processes. Leachates from landfills in the third and fourth phase generally require both biological and physical-chemical treatment.

This research evaluates problems encountered in a conventionally designed activated sludge plant receiving leachate. For this research, the biologically degradable high organic strength leachate produced from a landfill in the first and second phase was selected. This leachate is characterized by COD's in the range of 20,000 mg/l with acetic and propionic acid the major components. Amino acids and compounds containing the carbonyl group are also present.

EXPERIMENTAL APPROACH

The experimental apparatus consisted of two parallel 40 liter per minute activated sludge processes, a leachate storage tank, and the associated support equipment. One process functioned as the control system, the other as the experimental system. A predetermined amount of leachate was metered into the experimental system and the process allowed to equilibrate. After the stabilization period the processes were monitored at a quasi steady-state for four days, after which time the leachate dose was increased.

Experimental Apparatus

Both activated sludge systems were designed using the guidelines from "10 state standards" (3). Each of the activated sludge units contained three completely mixed passes as shown in Figure 1. The liquid volume of each pass was 6030 liters. Because the objectives of this research were to evaluate the conventional design and operating parameters of the activated sludge process, particular attention was given to process control. The air flow capacity of the aeration system for each pass was designed with a safety factor of two. This safety factor was to be

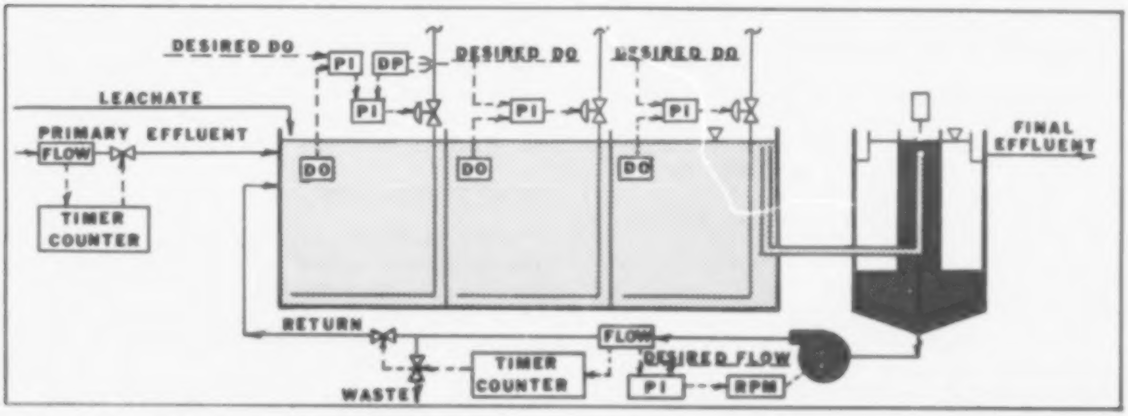

FIGURE 1. Process control diagram

evaluated as the potential limiting parameter in treating leachate. For this reason, the dissolved oxygen was controlled by two cascade PI control loops. The outer control loop monitored the dissolved oxygen in the aerator and determined the desired air flow. The inter control loop adjusted the air flow control valve to deliver the desired air flow. This control system maintained the dissolved oxygen in the mix liquor at 2.0 ± 0.1 mg/l (5% accuracy).

The sludge return control system was designed to be adjustable from 10 to 40 liters per min. (25% to 100% of main flow). The pumping system was designed to deliver the desired flow ± 0.5 liter per minute (1% accuracy). This was accomplished using a P.I. control loop which monitored the actual return flow, desired return flow, and determined the pump RPM.

The primary effluent flow (influent to the activated sludge process) was controlled by opening the influent valve to the first pass every 30 seconds, allowing 20 liters to flow into the first pass, and then closing the influent valve. This procedure was found to deliver 40 ± 0.1 liter per minute (0.2% accuracy) to each of the two parallel units.

This control system was implemented using a Texas Instruments process controller model number PM 550. The control system was found to satisfactorily maintain parallel operation of the two activated sludge processes during a six month shakedown period.

The operating parameters of the activated sludge process were as follows:

Sludge Return	10 liter per min.
SRT	1.5 day
Aeration Time	6 hours (including sludge recycle)
Secondary Clarifier Hydraulic Loading	15 m^3 per m^2 day

Synthetic Leachate

The first series of studies were conducted using a synthetic leachate. The synthetic leachate was designed to simulate the volatile acids, proteins, oxidation reduction potential, COD, and pH of sanitary landfill leachate. The following mixture was used.

0.15 M	sodium acetate
0.15 M	acetic acid
0.05 M	glycine
0.008 M	pyrogallol

The sodium acetate and acetic acid simulated the volatile acid and COD, and buffered the pH at 5. The glycine simulated the proteins and amino acids. The pyrogallol simulated the oxidation-reduction potential. the COD of this mixture was 22,000 mg/l which simulated a strong leachate produced from a young (1 to 5 yr.) landfill. This mixture was spiked into the 1st pass aerator of the experimental system at 100, 200, and 400 mg/l COD. Each spike level was run for one week. This provided several days for the hydraulic and biological systems to readjust and four days of quasi-steady -state operation.

Sampling and Lab Analysis

Samples were collected from the mix liquor and return activated sludge on four hour intervals and analyzed for centrifuge volume, settling volume, and oxygen uptake rates. Two-day composite samples were analyzed for TKN, NH_3, NO_3, Total P, COD, TOC and VSS.

CONCEPTUAL DATA ANALYSIS

The major evaluation parameters are the aeration requirements, specific oxygen utilization rate, sludge production, and sludge settling characteristic. Laboratory studies (2) indicated that sludge bulking occurred at high organic loadings. However, air flows and dissolved oxygen levels in laboratory units typically do not simulate full-scale units. Thus, the experimental design required close monitoring and control of the above parameters to simulate full-scale operation.

Aeration requirements

The aeration requirements to maintain the dissolved oxygen at 2 mg/l was monitored by orifice plates and differential pressure transducers. The actual air flow was compared to the maximum capacity air flow.

Specific Oxygen Utilization Rate

The specific oxygen utilization rate (SCOUR) was computed from the oxygen uptake rate of the mix liquor and return sludge divided by the respective volatile suspended solids of the sample. This value represents the oxygen utilization rate per gm of active bacteria. In theory, this value would be high in the mix liquor as the bacteria decompose the organics and low in the return sludge as the bacteria undergo endogenous respiration. Further, the SCOUR of the mix liquor in both the experimental and control system would remain the same as the organic load increased in the experimental system. This is because the strain at bacteria would remain the same. Only the bacteria mass would change. Similarly, the SCOUR of the return sludge would remain the same.

Sludge Production

The mass of sludge produced per mass of COD applied to the system should remain the same.

Sludge Settling Characteristics

The laboratory studies indicated that sludge bulking occurred at high organic loadings. This research monitored the sludge mass flow rate to the clarifier and the sludge volume index (SVI) to determine if the sludge mass, or SVI, or both caused bulking in the clarifier. The monitoring would also determine at what organic loading the bulking occurred.

DATA ANALYSIS

The data analysis consisted of plotting the five point moving average of the laboratory analysis conducted every four hours.

Figures 2 and 3 illustrate the specific oxygen utilization rate (SCOUR) of the mix liquor and return activated sludge for the control system and spike system. The SCOUR in these plots are expressed as mg of oxygen utilized per hour by one gm of VSS. The VSS, for these figures, were estimated from the centrifuge volume. Figures 2 and 3 illustrates that the SCOUR in the spike system and control system generally paralled each other. The exceptions were in the mix liquor following an increase in leachate spike. After the leachate spike was increased the biomass responded with an increase in growth rate which resulted in increasing the SCOUR to 70 mg/gm-hr. After about 40 hours of growth the SCOUR returned to the normal range of 40-50 mg/gm-hr. Figure 3 illustrates that the SCOUR of the return activated sludge ranged between 20-25 mg/gm-hr. This indicates that the return sludge was stabilized thus demonstrating that the aeration capacity was not exceeded. These two figures illustrate that the conventional design parameters for aeration capacity and mix liquor retention time are capable of stabilizing a leachate spike up to 400 mg/l COD.

Figure 4 presents the sludge volume index (SVI) of the control and spiked systems. Just after the 100 mg/l COD spike was started the SVI of the spiked system increased to 350 ml/gm. This was as expected due to the increase in organic loading. The SVI was then seen to decline during the 200 mg/l and 400 mg/l COD spikes. This indicated that the spiked system was adjusting to the organic loading and the sludge settling characteristics were improving. However, during the 400 mg/l COD spike the mass of bacteria within the spiked system increased as shown by the centrifuge volume in Figure 5. This increase in solids offset the improvement in sludge settling. The result was an uncontrolled increase in sludge volume as shown in Figure 6. At 470 hours the sludge volume filled the clarifier and flowed over the weirs thus "bulking" the clarifier.

Table 1 presents the average value of the control system and the average values, after the first 40 hours, of the three spike runs for various parameters. The data verify that the bacterial mass (VSS) increased in response to the organic loading (Influent COD) to maintain a food to microorganism ratio of 2.4 to 2.5 gm COD-applied/gm VSS-day. The oxygen uptake rate of the mix liquor increased from 30 mg O_2/l-hr in the control system to 75 mg O_2/l-hr in the 400 mg/l COD spike system. This increase did not overload the aeration system, however, the aeration system was observed to be at maximum capacity.

RESULTS

These three short runs were performed to determine what problems would occur in the activated sludge process when it is organically overloaded by sanitary landfill leachate. The activated sludge process was found to treat a synthetic leachate spike of 100 and 200 mg/l COD but not 400 mg/l COD. The process "bulked" at 400 mg/l COD

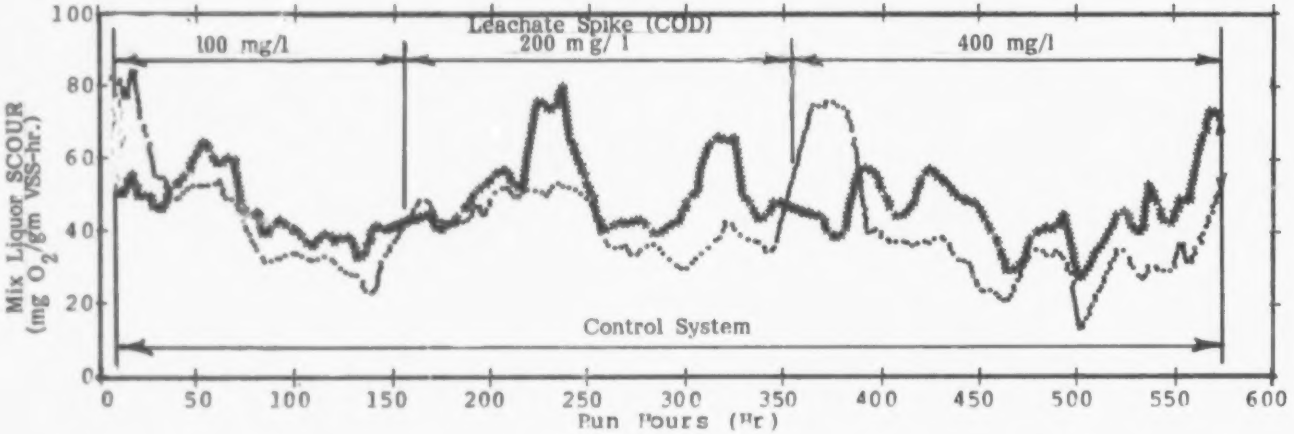

FIGURE 2. Mix liquor SCOUR with and without leachate spike

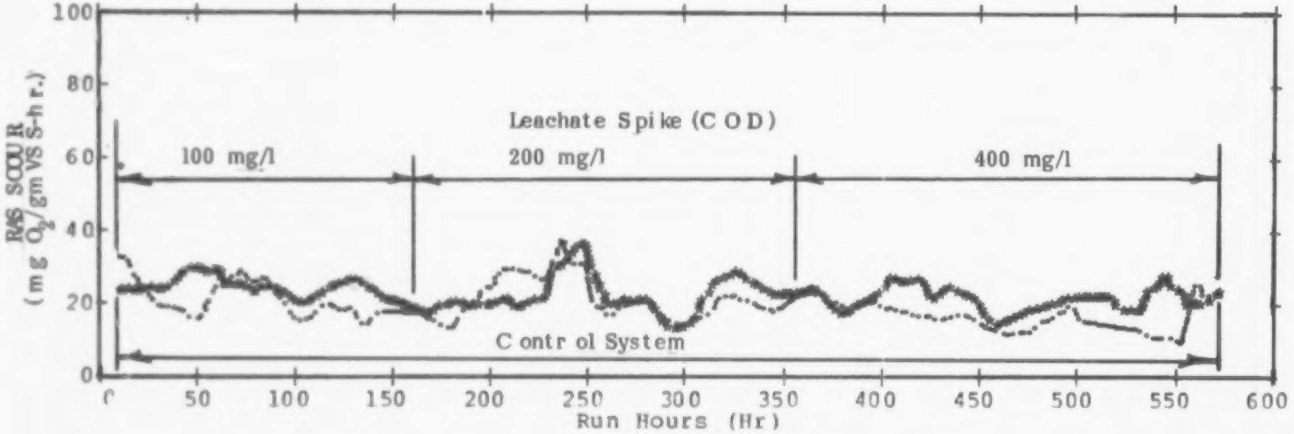

FIGURE 3. RAS SCOUR with and without leachate spike

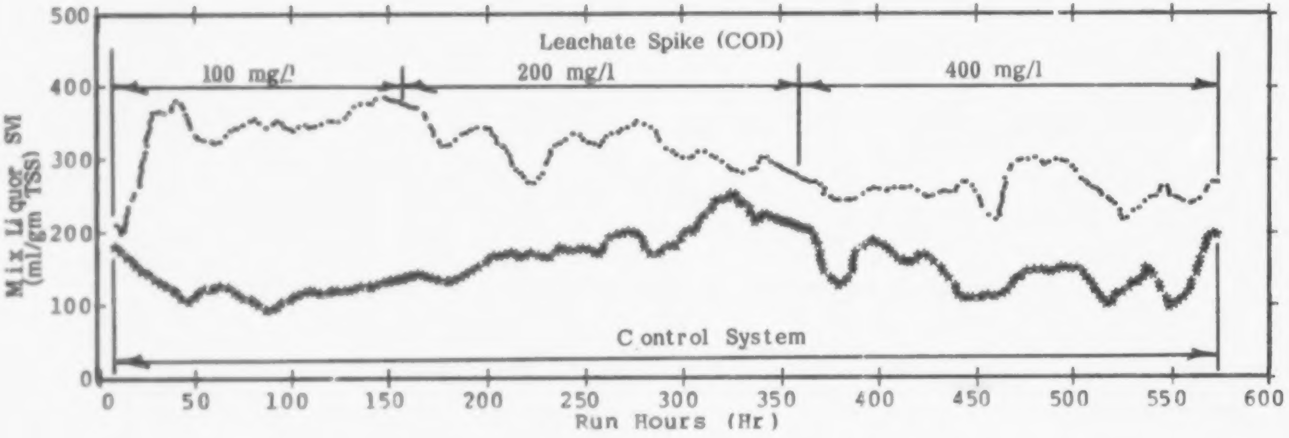

FIGURE 4. Mix liquor SVI with and without leachate spike

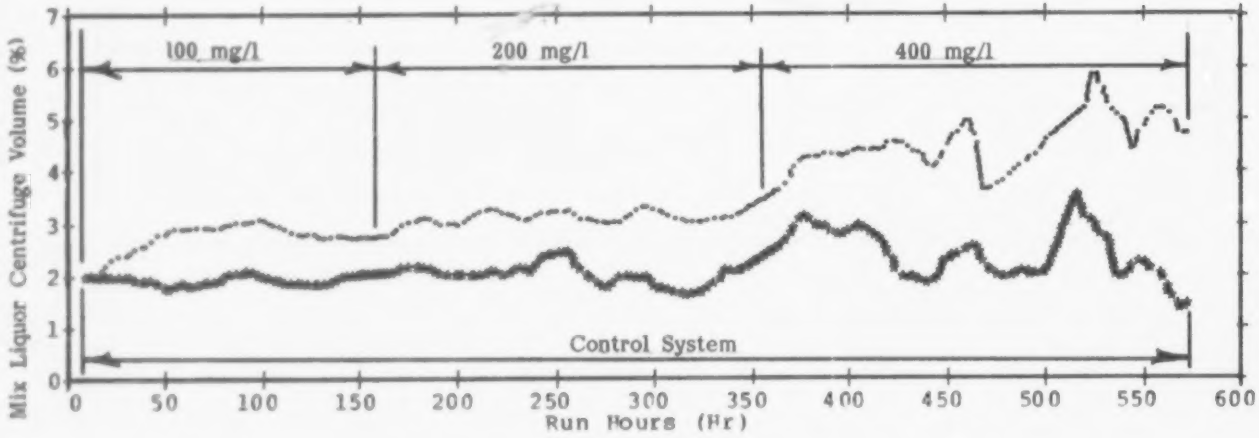

FIGURE 5. Mix liquor centrifuge volume with and without leachate spike

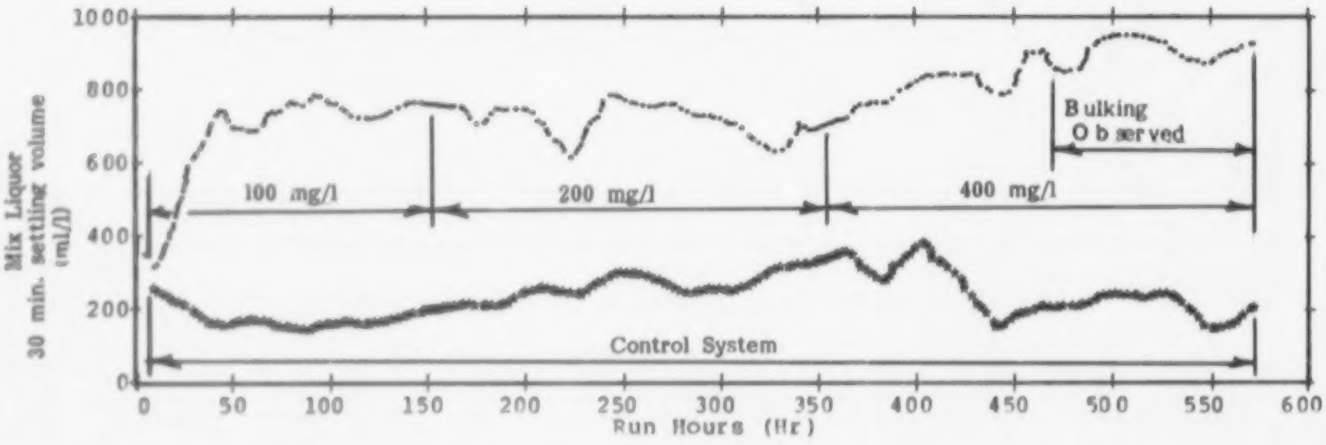

FIGURE 6. Mix liquor settling volume with and without leachate spike

Table 1. Analysis of Leachate Spikes

		Control	Leachate Spike (mg/l COD)		
			100	200	400
Influent:	COD (mg/l	670	700	870	1070
	TSS (mg/l)	290	290	290	290
Mix Liquor:	TSS (gm/l)	1.55	1.68	1.94	2.28
	VSS (gm/l)	0.85	1.00	1.15	1.45
	Non VSS (gm/l)	0.70	0.68	0.79	0.83
	Cent. Volume (%)	2.0	2.8	3.1	4.5
	Settling Volume (ml/l)	215	735	705	877
	O_2 Uptake (mg/l-hr)	38	46	55	75
	SVI (ml/gm)	140	440	360	385
	SCOUR Mg O_2/gm VSS-hr)	45	46	48	52
f:M (gm COD App/gm VSS-day)		2.5	2.4	2.4	2.4
Solids Production (gm TSS/gm COD App)		0.4	0.4	0.4	0.4
RAS:	TSS (gm/l)	6.6	8.0	8.4	10.9
	VSS (gm/l)	3.6	4.8	5.0	7.0
	NON VSS (gm/l)	3.0	3.2	3.4	3.9
	Cent. Volume (%)	9.5	14	17	25
	O_2 Uptake (mg/l-hr)	80	115	150	180
	SCOUR (mg O_2/gm VSS-hr)	22	24	30	26
RAS TSS:ML TSS Ratio		4.2	4.8	4.3	4.8
Clarifiers:					
Solids Loading (Kg/M^2-day)		29	31	36	42
Sludge Volume Loading (m^3/m^2-day)		4.0	14	13	16
Sludge Volume Flow:					
	To Clarifier (% Q)	27	92	88	110
	From Clarifier (RAS + WAS)(%Q)	30	30	30	30
Effluent:	COD (mg/l)	108	135	85	200
	TSS (mg/l)	14	10	8	86

spike due to an increase in sludge volume. The major problem was a combination of increase in sludge mass and degradation in sludge settling characteristics that resulted in the sludge volume overloading the clarifier.

Sludge Settling Characteristic

The sludge settling characteristic were monitored using SVI. Upon starting the initial 100 mg/l COD spike the SVI increased from the normal 100 to 200 ml/mg range to 350 ml/gm. The SVI was seen to improve throughout the 560 hour test period. The length of this test period was not sufficient to restablize the SVI. Thus, the higher loading may have been treatable if the activated sludge system was given time to acclimate at the lower organic loadings.

Aeration Requirements

The oxygen uptake rate at the mix liquor in the 1st pass of the experimental system increased to 75 mg O_2/l-hr compared to 38 mg O_2/l-hr in the control system. The conventional design for an aeration systems treating Cincinnati Mill Creek Sewage was capable of meeting this increase requirement. However, aeration systems designed to treat a lower strength sewage than the above may not have the reserve capacity. The increase in oxygen uptake rate was seen to be 10 mg O_2/l-hr per 100 mg/l COD spike. This increase in oxygen uptake rate should be considered in evaluating other aeration systems.

Specific Oxygen Utilization Rate

The specific oxygen utilization rate (SCOUR) was seen to remain relatively constant in the mix liquor and return sludge. The SCOUR generally ranged from 40 to 50 mg O_2/gm VSS-hr in the mix liquor and ranged from 20 to 25 mg O_2/gm VSS-hr in the return sludges.

Solids Production

The mass of solids produced per COD applied was seen to remain unchanged at 0.4 gm TSS/gm COD as the organic loading increased.

RECOMMENDATIONS:

The following topics require further research:

- Determine if the SVI does recover if the activated sludge system in given time to acclimate.
- Continue the above research using leachate from a sanitary landfill.
- Evaluate different process control strategies in treating the high organic loadings.

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CONTAINMENT OF HEAVY METALS IN LANDFILLS WITH LEACHATE RECYCLE

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ABSTRACT

The presence, transport and ultimate fate of heavy metals in landfill leachate are discussed with respect to the opportunities for physical-chemical interaction within the landfill environment. Most leachates were found to contain high concentrations of dissolved ions and a broad array of organic and inorganic ligands which, coupled with the anaerobic reducing conditions of the landfill, favored the removal of heavy metals primarily by precipitation as sulfides. This removal was enhanced by the increased stabilization rates and filtering action promoted by the recirculation of leachate through the landfill mass.

These observations were fortified by data obtained from analysis of leachate samples from four simulated landfill lysimeter columns operated under conditions of leachate collection and recycle and with codisposal of residential-type refuse and metal plating sludge. Utilizing equilibrium concepts modified by considerations for ionic strength and by activity corrections, the potential for precipitation and/or complexation of heavy metals and the profound impact of sulfides on these reactions were described.

INTRODUCTION

Most landfills receiving waste materials and an influx of moisture will provide unusual chemical environments for generation of leachate containing high levels of dissolved, ionic species and a complex array of organic compounds containing a variety of functional groups. In addition, depending on a number of factors, leachates can exhibit highly oxidizing or highly reducing tendencies which may further influence the potential hazard of heavy metal contaminants. Since heavy metals such as copper (Cu), cadmium (Cd), nickel (Ni) and mercury (Hg) are considered priority pollutants, knowledge of their chemical behavior and relative mobility or reactivity in landfill leachates is of major concern. This paper will provide additional data and evaluation concerning the influence of chemical activity and complexation on the behavior of heavy metals in landfill leachates.

LEACHATE SOURCE

While a small amount of data derived from literature reports of other investigators have been analyzed herein, the bulk of the data have been obtained from studies by the authors. The leachate used in this evaluation was obtained from four simulated landfill lysimeter columns being used to study the codisposal of metal plating sludge with municipal refuse under the influence of leachate containment and recycle. Accordingly, one column contained residential-type refuse, the other three, refuse plus varying amounts of sludge containing zinc (Zn), chromium (Cr), cadmium (Cd) and nickel (Ni) from a local electroplating industry. The columns are operationally similar to the units indicated in Figure 1 and have been described in more detail elsewhere with some preliminary observations regarding metal solubility and oxidation state conditions [1,2]. Operation of these lysimeters with leachate recycle

TABLE 1. SELECTED LEACHATE CHARACTERISTICS FROM LYSIMETER COLUMNS*

Parameter	Analyses			
	Column 1	Column 2	Column 3	Column 4
pH	7.0	6.2	6.2	6.4
Conductivity, μmho cm^{-1}	2,900	4,000	6,000	5,100
Chlorides, mg/ℓ	420	470	520	680
COD, mg/ℓ	890	5,120	8,100	5,400
TOC, mg/ℓ	300	1,800	2,000	1,900
Ortho-Phosphorus, mg/ℓ	7	3	6	6
TKN, mg/ℓas N	60	100	80	90
Ammonia, mg/ℓ as N	6	6	6	5
Sulfide, mg/ℓ	0,9	0.9	0.9	0.8
ORP, E_c, mV	-240	-120	-70	-60
Cd, mg/ℓ	0.1	0.2	0.6	1.3
Ni, mg/ℓ	0.1	0.4	0.5	0.8
Volatile Acids, mg/ℓ as CH_3COOH	250	4,100	3,800	4,800
Total Carboxyl, mg/ℓ as CH_3COOH	1,400	6,500	7,600	3,200
Aromatic Hydroxyl, mg/ℓ as Tannic Acid	40	60	80	70
Amino Acids, mg/ℓ as Leucine	200	500	700	800

* Sample obtained 300 days after loading the columns.

Column 1: Control without metal sludge.

Column 2: Test unit with 33.6 kg of metal sludge.

Column 3: Test unit with 65.8 kg of metal sludge.

Column 4: Test unit with 135.2 kg of metal sludge.

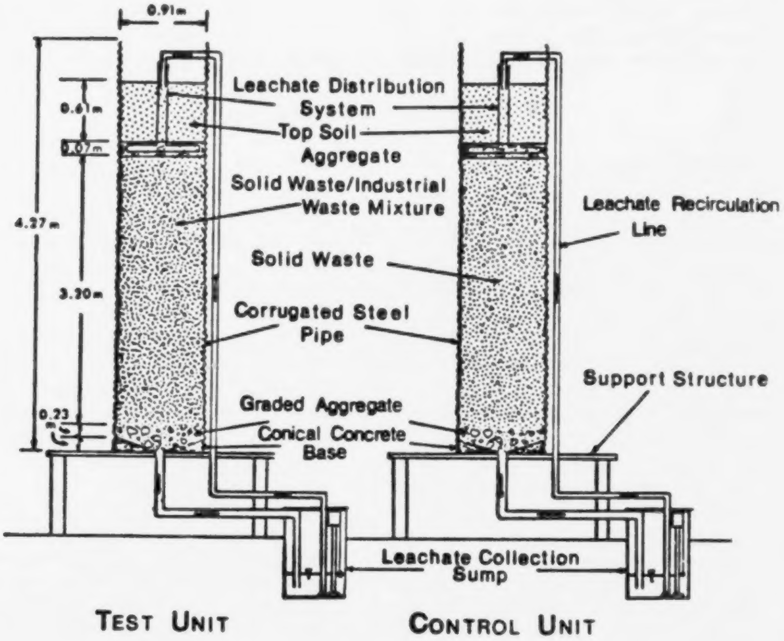

0.91m
0.61m
0.07m
4.27m
3.20m
0.23 m
Leachate Distribution System
Top Soil
Aggregate
Solid Waste/Industrial Waste Mixture
Solid Waste
Corrugated Steel Pipe
Graded Aggregate
Conical Concrete Base
Leachate Recirculation Line
Support Structure
Leachate Collection Sump
TEST UNIT
CONTROL UNIT

Figure 1. Simulated Landfill Lysimeter Columns with Leachate Recycle.

has been continued during the past year and leachate samples have been analyzed for inorganic constituents as well as several of the organic functional groups.

ANALYTICAL METHODS AND DATA

The leachate from the columns has been analyzed for inorganic components by conventional methods. Metals have been analyzed by digestion and atomic absorption spectroscopy, chloride and sulfide by electrochemical techniques, nutrients by Autoanalyzer methods, and ORP, pH and conductivity by the usual instrumental methods. Analysis of organic constitutents included direct aqueous injection gas chromatography for volatile organic acids, the photometric ninhydrin method for amino acids[3], the dinitrophenylhydrazine method for carbonyl compounds[4], the folin-Dennis method for aromatic hydroxyl groups[5] and the hydroxylamine colorimetric method for carboxyl groups[6]. A typical range of analytical results for this leachate after 300 days since loading the columns is given in Table 1; overall changes in selected analytical results are indicated in Figures 2 through 8.

The equilibrium constants used for analysis herein have been compiled from Sillen [7,8]. Pertinent constants for the various inorganic species chosen are summarized in Table 2.

CHEMICAL ACTIVITY

Chemical activity is, in effect, a measure of apparent rather than actual ion concentrations. Moreover, representing as it does the result of interactions among dissolved ionic species, activity is a function of the ionic strength of the solution and is expressed as dimensionless activity coefficients. These coefficients, which represent that fraction of the actual ion concentration which exists as active free ions, are calculated by means of several semiempirical equations and are invariably less than one. Therefore, the activity coefficient provides a measure of the extent to which an ionic species will participate in a given chemical reaction. Obviously, if an activity coefficient deviates significantly from unity, simple concentration-based chemical calculations will be seriously in error.

Typical activity coefficients in nonsaline natural waters and many dilute wastewaters tend to be sufficiently near unity that the impact of activity on chemical behavior in these systems can generally be neglected. By contrast, the high dissolved solids content of most landfill leachate results in activity coefficients substantially smaller than unity particularly in the case of polyvalent ions. Therefore, it is appropriate to explore the effect of ionic strength on the chemical reactivity of leachate constituents.

Computation of Ionic Strength in Leachates

Computation of activity coefficients is dependent on a knowledge of the ionic strength of the medium involved. Classically the ionic strength is calculated on the basis of a comprehensive qualitative and quantitative inventory by means of Equation 1.[9]

$$\mu = \tfrac{1}{2} \sum_i (C_i Z_i^2) \qquad (1)$$

where: μ = ionic strength
C_i = concentration of species, i
Z_i = charge of species, i

Such a comprehensive inventory of ion concentrations is relatively simple to obtain in natural and drinking waters as well as in some wastewaters. Landfill leachates, on the other hand, present a far less accommodating matrix in which to carry out the necessary analyses. Such factors as high levels of competing solutes, intense color and lack of any complete qualitative description of leachate composition renders the foregoing method of ionic strength determination subject to very significant levels of uncertainty. (Indeed, high levels of ionic strength will tend to make measurements such as those involving specific ion electrodes suspect). Therefore, the work reported herein has been based on well established correlations between ionic strength and the easily measured property of electrolytic conductance.[10-12].

The linear relationship between ionic strength and conductivity, $\mu = 1.6 \times 10^{-5} \times$ conductivity, demonstrated independently by Lind[10] and by Russell[12], was used with leachate conductivity data collected over a period of nearly 400 days. The resulting

TABLE 2. SELECTED EQUILIBRIUM CONSTANTS FOR INORGANIC SPECIES

Species	Constant	Log of Value
H_2S	K_{a_1}	- 7.0
H_2S	K_{a_2}	-13.7
NH_4^+	K_a	- 9.3
NiS	K_{s_o}	-23.8
Ag_2S	K_{s_o}	-50.2
Hg_2S	K_{s_o}	-52.5
$Hg_2(OH)_2$	K_{s_o}	-23.2
$Cu_3(CO_3)_2(OH)_2$	K_{s_o}	-45.4
$Cr(OH)_3$	K_{s_o}	-30.5
$AgCl_2^-$	β_2	+ 5.2
$HgCl_4^{-2}$	β_4	+15.9
Hg_2Cl_2	β_2	+12.3
$Cu(NH_3)_4^{+2}$	β_4	+13.0
$Hg(NH_3)_2^{+2}$	β_2	+17.8
$Hg_2(NH_3)_2^{+2}$	β_2	+17.5
$Zn(NH_3)_4^{+2}$	β_4	+ 9.3

Note: β_n is the constant for the reaction between a metal, M, and a ligand, L, or $M + nL \quad ML_n \quad K = \beta_n$

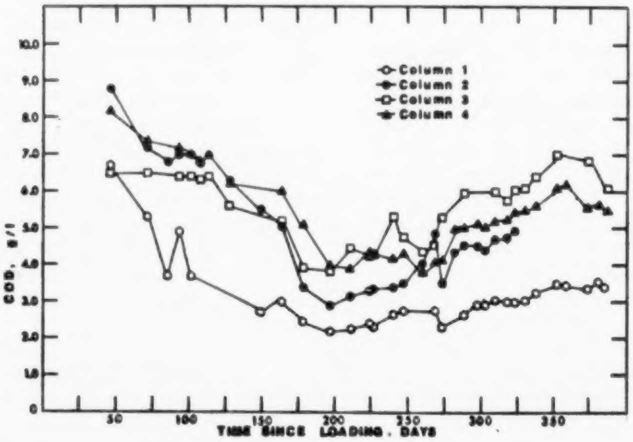


Figure 2. COD as a Function of Time since Loading Columns.

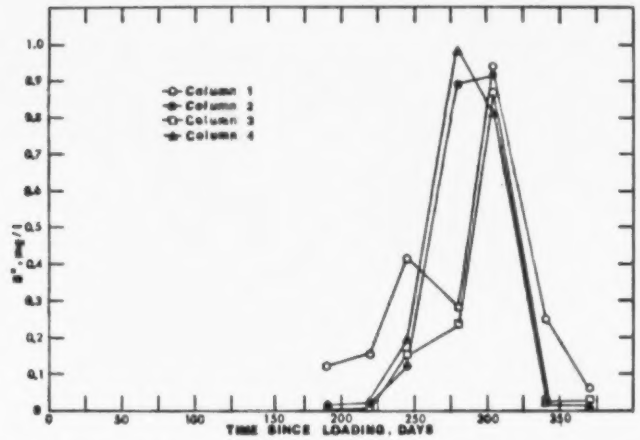


Figure 3. Sulfide Concentration as a Function of Time since Loading Columns.

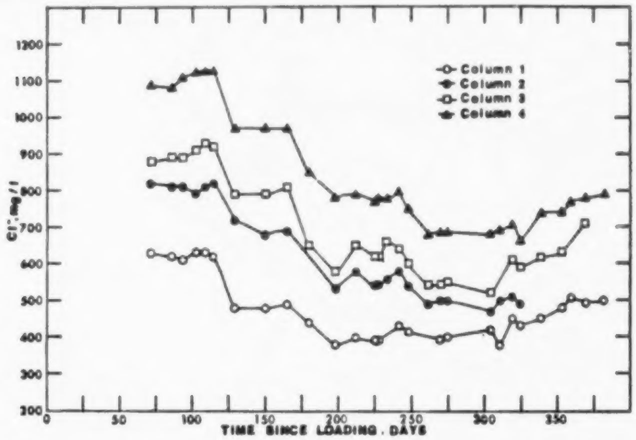


Figure 4. Chloride Concentration as a Function of Time since Loading Columns.

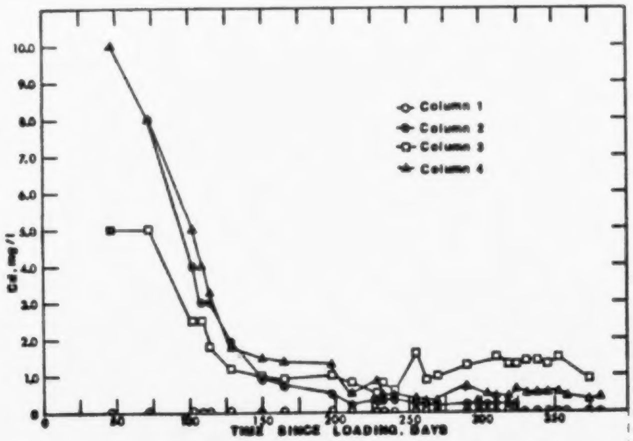


Figure 5. Cadmium Concentration as a Function of Time since Loading Columns.

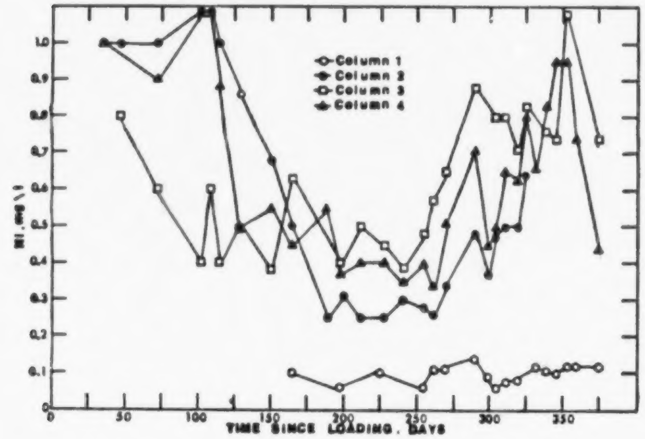


Figure 6. Nickel Concentration as a Function of Time since Loading Columns.

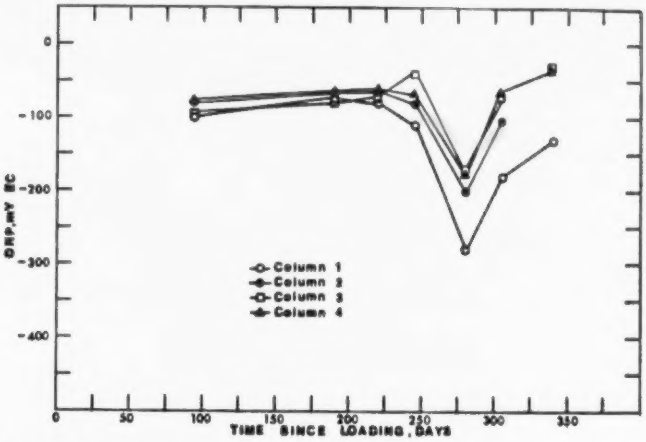


Figure 7. Oxidation Reduction Potential as a Function of Time since Loading Columns.

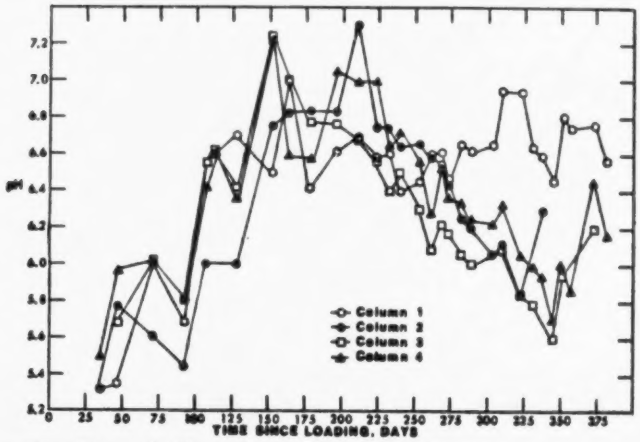


Figure 8. pH as a Function of Time since Loading Columns.

ionic strengths are plotted in Figure 9. The data for Columns 2 through 4, containing the metal sludge, have been averaged since no obvious trends of differences were apparent between them. Therefore, it was concluded that the resulting ionic strength data, while certainly not as precise as those which might be calculated on the basis of exact analytical determinations, were probably subject to less error than might be expected from any specific analyses in the complex leachate matrix.

Further examination of Figure 9 shows some interesting trends which will be reflected in the calculated activity coefficients. In all columns, the trend of leachate ionic strength with time followed the same pattern; high initial values were followed by a steady decrease to minimum values 30-50 percent of the initial value after about 200 days. Following this minimum, a slow increase in ionic strength was observed with recovery of a substantial fraction of the decrease of the ionic strength by Day 400. For Columns 2 through 4, there was a further suggestion of a decrease in the ionic strength after Day 350; this decrease did not appear for Column 1.

While an unequivocal explanation for the observed trends cannot be advanced, it seems likely that the initial decrease in μ was a response to the washout of easily mobilized ions such as sodium (Na^+), chloride (Cl^-) and sulfate (SO_4^{-2}) combined with such factors as the conversion of sulfate to sulfide under increasingly reducing conditions consequenced by anaerobic biological activity. The subsequent precipitation of the sulfide as heavy metal sulfides would tend to withdraw significant ionic strength from solution. The increase in ionic strength after 200 days may have been due to some reduction in leachate volume due to evaporation and/or the gradual release of less mobile ions from the landfill materials as they decomposed. In addition, the generation of volatile carboxylic acids by biological activity at pH levels over six would also add anions and contribute to the ionic strength. Other products of the decomposition of organic matter such as amino acids and phenolic compounds probably also contributed to the rising trend in ionic strength.

The late decrease in leachate ionic strength in Columns 2 through 4 may have been the result of a corresponding decrease in pH as shown in Figure 8. This decrease became especially pronounced after 250 days. While the hydrogen ions make a negligible direct contribution to μ at pH values higher than four, by altering the distribution between the ionized and neutral forms of acids and bases, small changes in the concentration of hydrogen ions will be reflected in major changes in μ. In this specific case, the observed decrease in pH from $\simeq$6.2 to $\simeq$5.5 when applied to a total volatile acid concentration of 2,200 mg/ℓ as acetic acid is sufficient to account for 30 percent of the decrease in μ based simply on conversion of acid anions to neutral acids. Similar pH dependant transformations of other less fully defined acidic species might well have accounted for the remainder of the decrease in μ. It is interesting to note that Column 1, which experienced no important decrease in leachate pH after about 200 days, also experienced no late decrease in ionic strength.

Another factor likely having an important impact on ionic strength is biological activity in the columns. An examination of Figure 9 indicates that, the control column showed a smooth decrease in ionic strength to a stable level after about 160 days, suggesting that biological activity had largely ceased in Column 1. On the other hand, Columns 2 through 4 displayed an initial decrease in ionic strength followed by an apparent retardation between 60 and 160 days. This period of retardation probably reflects a corresponding inhibition of biological activity by the heavy metal sludge. The increase in ionic strength in Columns 2 through 4 after 200 days probably reflects to some extent an upswing in biological activity resulting in increased concentrations of acid anions in the leachate. Finally, increased acid production in Columns 2 through 4 leads to the very late drop in pH and corresponding decrease in ionic strength.

Examination of the ionic strength data yielded little concrete evidence of systematic trends among the columns to which the metal sludge was added. All three of these columns produced leachate with ionic strength values which were consistently significantly higher than in

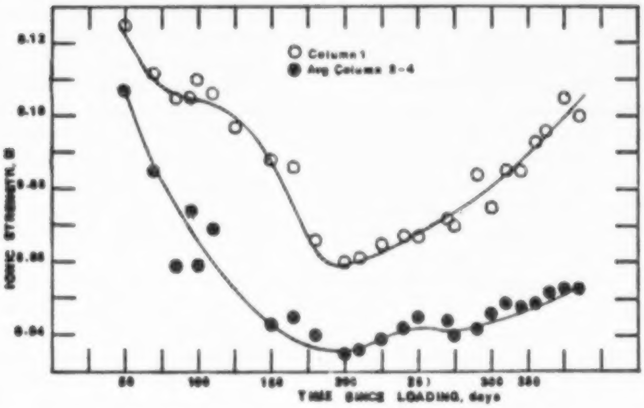


Figure 9. Ionic Strength as a Function of Time since Loading Columns.

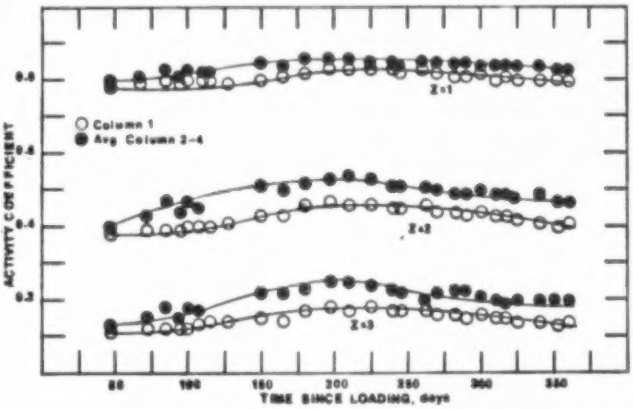


Figure 10. Activity Coefficients for Mono-, Di- and Trivalent Ions as a Function of Time since Loading Columns.

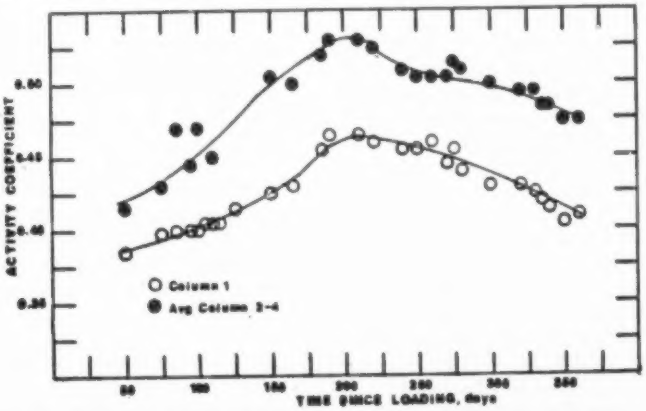


Figure 11. Activity Coefficients for Divalent Ions as a Function of Time since Loading Columns.

the control column. This was expected due to the addition of ionic contributions originating with the waste metal plating sludge.

Estimation of Activity Coefficients in Leachate

The estimation of activity coefficients, especially in systems of very high ionic strength, presents special difficulties. The most common expression for activity coefficients, the DeBye-Hückel equation[9] as given in Equation 2 applies only up to values of μ of 5×10^{-3} M.

$$-\log \gamma_i = 0.5\, Z_i^{\,2}\, \mu^{1/2} \qquad (2)$$

where γ_i = the activity coefficient of ion, X_i

Z_i = the charge of the ion

μ = the ionic strength

Since leachates usually have ionic strengths well in excess of 5×10^{-3} M, this equation is inapplicable for such systems. However, two extended approximations are available which afford estimates of γ at substantially higher values of μ. These are the extended DeBye-Hückel expression[9] given in Equation 3 and usuable to μ values of approximately 0.1 - 0.15 M,

$$-\log \gamma_i = \frac{0.5\, Z_i^{\,2}\, \mu^{1/2}}{1 + \mu^{1/2}} \qquad (3)$$

and the Davies [9] expression given in Equation 4 and applicable up to $\mu \simeq 0.6$M.

$$-\log \gamma_i = 0.5 Z_i^{\,2} \left(\frac{\mu^{1/2}}{1 + \mu^{1/2}} - 0.3\mu \right) \qquad (4)$$

Since, in the system under study, the ionic strengths measured in the leachates were consistently near or below 0.1 M, the extended DeBye-Hückel expression was used for calculation of γ_i. Values of γ_i for ions of charge one, two and three are shown as a function of time in Figure 10. Figure 11 shows the trends of γ as a function of time for divalent ions on a more detailed scale.

As is shown in Figure 10, the activity coefficients deviate markedly from unity. Thus, ions with a charge of one have coefficients of approximately 0.83, divalent ions have γ values of about 0.43 and trivalent ions have coefficients in the range of 0.1 to 0.2. It is also apparent from Figure 10 that γ is quite insensitive to variations in leachate composition.

Figure 11, an expanded scale plot of activity coefficient data for a divalent ion, provides a more detailed representation of the trends which do occur.

The activity coefficients display the expected inverse relationship to ionic strength as shown in Figure 9.

Impact of Activity Corrections on Heavy Metal Solubilities

The implications of the observed values of the activity coefficients toward the solubility of heavy metals in leachate are substantial. For example, it has been demonstrated that the solubility of trivalent chromium in leachate is controlled by hydroxide equilibria.[2]

$$[Cr^{+3}]\,[OH^-]^3 = K_{so} \qquad (5)$$

This expression will be modified by activity coefficient considerations to

$$\gamma_{Cr^{+3}}[Cr^{+3}]\ \gamma^3_{OH^-}\ [OH^-]^3 = K_{so} \qquad (6)$$

or,

$$[Cr^{+3}][OH^-]^3 = K_{so} / \gamma_{Cr^{+3}}\, \gamma^3_{OH^-} \qquad (7)$$

Applying values of activity coefficients reasonably representative of the metal sludge laden columns as shown in Figure 10, i.e., $\gamma_{Cr^{+3}} = 0.15$ and $\gamma_{OH^-} = 0.83$, leads to the following corrected expression for the solubility of trivalent chromium.

$$[Cr^{+3}]\,[OH^-]^3 = 11.7\ K_{so} \qquad (8)$$

Thus, on the basis of these considerations, it would be expected that the true chromium concentration in these leachate systems would be higher than those predicted by concentration consideration alone by a factor of ≈ 12.

During periods when sulfides were present in the leachate samples as presented in Figure 3, sulfide solubilities would be expected to be similarly effected by activity considerations. If the C_t for the sulfide system is expressed as

$$C_t = [H_2S] + [HS^-] + [S^{-2}] \quad (9)$$

an equation for the concentration of sulfide corrected for activity will take the form

$$[S^-] = C_t\left(\frac{[H^+]\gamma_{S^{-2}}}{K_{a_2}} + \frac{[H^+]^2\gamma^2_{H^+}\ \gamma_{S^{-2}}}{K_{a_1}\ K_{a_2}} + 1\right)^{-1} \quad (10)$$

Again from figure 10, with $\gamma_{H^+} = 0.83$ and $\gamma_{S^{-2}} = 0.43$, the effect of activity on the actual sulfide concentration in leachate can best be illustrated by means of a pC-pH diagram. Figure 12 shows a pC-pH diagram comparing the concentration of sulfide ion as a function of pH when activity is included with the concentration calculated neglecting activity. As indicated, the sulfide concentration is consistently higher by a factor of almost four when activity corrections are made. In situations where metal ions are of concern, such increased availability of S^{-2} could dramatically influence their mobility and ultimate fate since the importance of the sulfide ion as a precipitant for most toxic metal ions has been clearly shown.[2]

Correcting metal sulfide solubilities for activity can also be approached in a similar fashion. Consider as an example the solubility of nickel in the presence of sulfide.

$$\gamma_{Ni^{+2}}[Ni^{+2}]\gamma_{S^{-2}}[S^{-2}] = K_{so} = 1.7 \times 10^{-24} \quad (11)$$

Again using $\gamma_{S^{-2}} = \gamma_{Ni^{+2}} = 0.43$

then,

$$[Ni^{+2}][S^{-2}] = 9.19 \times 10^{-24} \quad (12)$$

or,

$$pNi^{+2} = 23.04 - pS^{-2} \quad (13)$$

Using the corrected sulfide concentrations from Figure 12 in this expression yields the curve shown in Figure 13.

It should be noted that while activity corrections applied to the solubility product of nickel sulfide will result in an increase in the solubility the elevation of sulfide concentrations shown in Figure 12 has the effect of decreasing the nickel solubility. As a result of these opposing tendencies, the corrected as opposed to the uncorrected solubility diagrams correspond fairly closely under the conditions chosen. Other systems will have to be considered on a case by case basis.

Similary, under oxidizing rather than reducing conditions, where sulfide would tend to control the metal solubility, precipitation of copper as carbonates will be subject to major modification as a result of activity factors. For example, one major solid copper carbonate has the formula $Cu_3(CO_3)_2(OH)_2$. Using the activity coefficients employed previously, the solubility product for this species, corrected for activity,

$$\gamma^3_{Cu^{+2}}[Cu^{+2}]^3\gamma^2_{CO_3^{-2}}[CO_3^{-2}]^2\gamma_{OH^-}[OH^-] = K_{so} \quad (14)$$

would be higher by a factor of approximately 100. A factor of this magnitude might well have a major impact on the chemical behavior and mobility of copper ions in a landfill leachate.

While the values of conductance used herein are fairly typical of landfill leachates, other researchers have observed leachates with conductances approaching 20,000 μ mhos cm^{-1}. Conductances of this magnitude indicate ionic strength values which approach 0.3 $\underline{M}$ and values of γ of 0.74 for monovalent ions, 0.30 for divalent ions and 0.065 for trivalent ions. Constants of this magnitude would have a substantial impact on leachate chemistry. For example, the solubility of chromium as the hydroxide would be increased by a

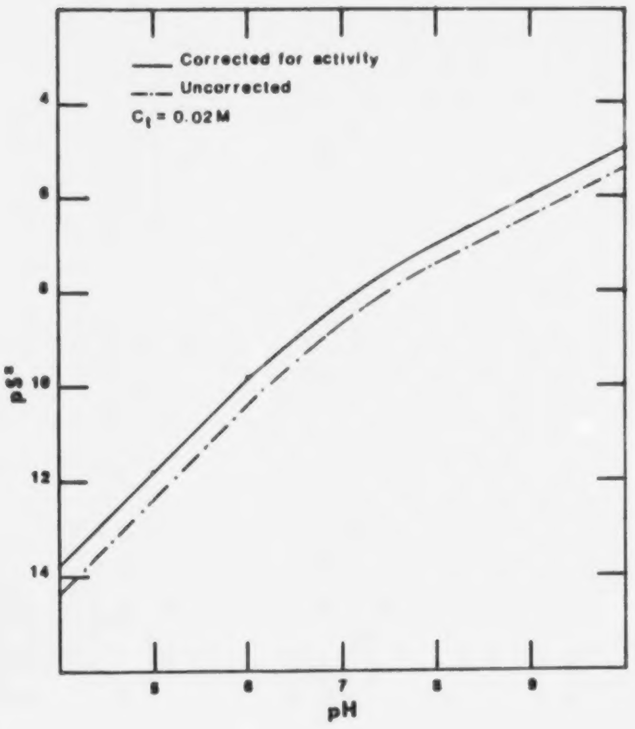


Figure 12. Concentration of Sulfide Ions as a Function of pH

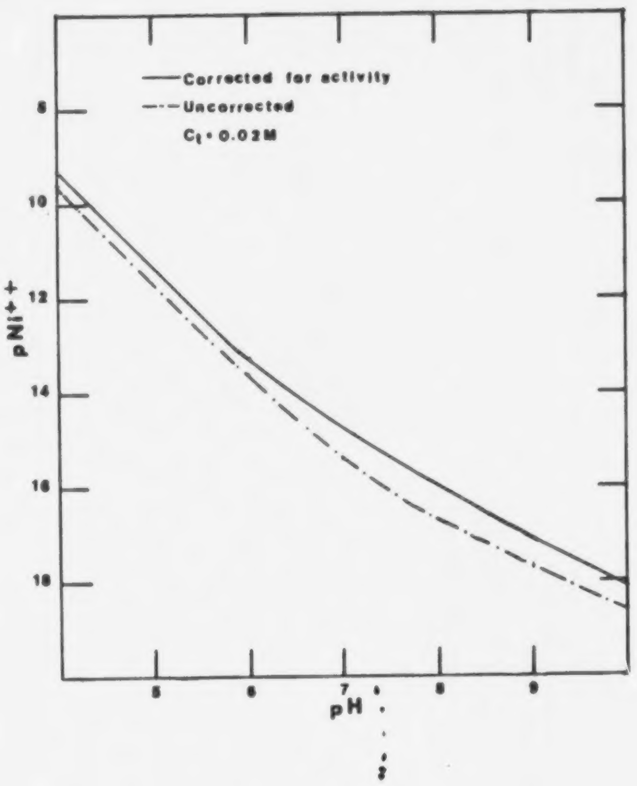


Figure 13. Concentration of Nickel in Solution in Equilibrium with Sulfide as a Function of pH

factor of approximately 40 times while the solubility product of $Cu_3(CO_3)_2(OH)_2$ would increase about 750 times. Thus, under the more extreme conditions which might exist in leachates, activity can be expected to distort simple concentration based equilibrium calculations substantially. Failure to recognize and compensate for the differences between activity and concentration in the analysis of leachates may lead to serious errors in evaluation and interpretation of leachate characteristics.

CHEMICAL COMPLEXATION

Heavy metals dissolved in aqueous systems exist not as the free ions but in combination with other chemical species in the form of complexes. In complexes, metal ions combine with non-metallic compounds called ligands by means of coordinate-covalent bonds. For example, a cupric ion forms a complex with four molecules of the ligand water as shown.

$$Cu^{+2} + 4H_2O \rightleftharpoons Cu(H_2O)_4^{+2}$$

While complexation in natural waters almost exclusively involves either water or the hydroxide ion as the ligand, leachates, being abundantly provided with such ligands as chloride, ammonia, phosphate and sulfate as well as an array of organic compounds provide conditions under which complexation must be considered in evaluating the transport and fate of toxic metal ions.

The complexation of heavy metals in landfill leachates is an area in which even more work remains to be done. Indeed, on a conceptual basis, it is easier at this point to eliminate possible complexing agents from positions of importance in determining heavy metal speciation. The greatest source of uncertainty is the rather incomplete knowledge regarding specific composition of leachates.

Inorganic Ligands

Some fairly reliable information is available on inorganic complexing agents. The most likely candidates as inorganic complexing agents in landfill leachates are chlorides, sulfates, phosphates and ammonia.

Effect of Sulfide

Of extreme importance to complexation is the impact of sulfide solubility equilibria on the levels of complex which can exist in the presence of even very small concentrations of sulfide. This is a consequence of the very low solubilities of heavy metal sulfides. In essence, sulfide very effectively competes with most complexing agents. Consequently, many heavy metals will precipitate as sulfides rather than remain in solution as complexes. For example, the concentration of the $Hg_2(NH_3)_2^{+2}$ complex in the absence of sulfide will be controlled by a reaction with a pK of 5.7

$$Hg_2(OH)_2(s) + 2NH_3 \rightleftharpoons Hg_2(NH_3)_2^{+2} + 2OH^-$$

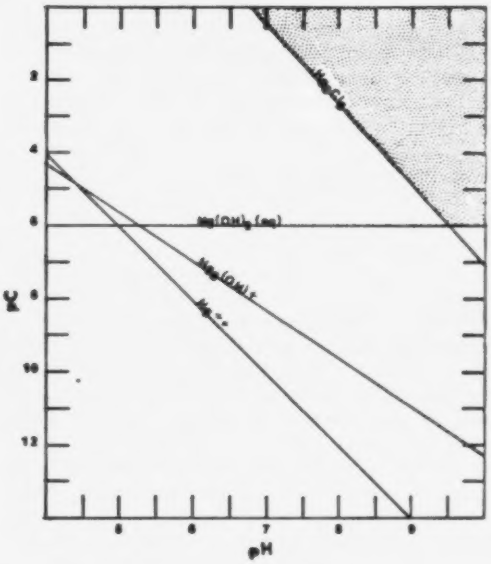
while in the presence of sulfide ions, the equivalent reaction will be

$$Hg_2S(s) + 2NH_3 \rightleftharpoons Hg_2(NH_3)_2^{+2} + S^{-2}$$

and have a pK of 34.9. Clearly, this reflects a massive difference in solubilities. Even at sulfide concentrations as low as 10^{-12} molar, $Hg_2(NH_3)_2^{+2}$ levels would be predicted to be on the order of 10^{-30} molar or less. Concentrations this low are virtually without meaning beyond suggesting that the rate at which precipitation occurs might control solubility of Hg_2^{+2} in the presence of sulfide ions. Otherwise, in those cases in which inorganic complexes might be significant, i.e., for Hg^{+2} and Hg_2^{+2}, the metal sulfide solubility equilibria are of such magnitudes that, in the presence of sulfide, metal complexation should be of little or no importance. Precipitation/filtration of the metals as the solid sulfides will control the metal behavior totally particularly where leachate recycle serves to accelerate these processes.

Chloride

While, as a general rule the chloride ion is a rather weak complexing agent, its effectiveness in complexing the two forms of mercury is quite high.



$Hg_2(OH)_2(s) \rightleftharpoons Hg_2^{+2} + 2OH^-$ pK = 23.2

$Hg_2(OH)_2(s) \rightleftharpoons Hg_2(OH)^+ + OH^-$ pK = 15.5

$Hg_2(OH)_2(s) \rightleftharpoons Hg_2(OH)_2(Aq)$ pK = 6.0

$Hg_2(OH)_2(s) + 2Cl \rightleftharpoons Hg_2Cl_2 + 2OH^-$ pK = 10.9

Figure 14: Distribution of Hg_2^{+2} Species as a function of pH in the presence of 0.019 M chloride (680 mg/l)

$Hg^{+2} + 4Cl^- \rightleftharpoons HgCl_4^{-2}$ $\log \beta_4 = 15.9$

$Hg_2^{+2} + 2Cl \rightleftharpoons Hg_2Cl_2$ $\log \beta_2 = 12.3$

Thus, in the absence of sulfide, solubility of Hg_2^{+2} will be controlled by the solubility of the neutral Hg_2Cl_2 species at pH values below about 9 in a system containing as much as 680 mg/l of chloride (Table 1). The resulting pC-pH diagram for this system is shown in Figure 14.

The shaded area of this graph represents the solubility region for Hg_2^{+2} in this leachate. As can be seen, below a pH of about 9 the major soluble species is the neutral dichloride complex Hg_2Cl_2. Thus, in this instance, chloride complexation is potentially of major significance in determining the behavior of a toxic metal.

Sulfate

Although sulfate is abundantly present in many landfill leachates, examination of the literature suggests that, relative to other available ligands, sulfate is simply too poor a complexing agent to be a major factor in heavy metal complexation.

It must be recognized however, that, while sulfate has very little importance as a direct complexing agent, its significance as a precursor for sulfide under the influence of anaerobic conditions and microbial activity cannot be minimized.

Phosphate

Polyphosphates tend to be very potent ligands for many heavy metals. However, it seems likely that phosphates will be of only minor importance in most landfill leachates. Following a rapid early discharge of high levels of phosphate with values approaching 30 mg/l of phosphorous, phosphorous levels in the control column leachate decreased rapidly to values in the vicinity of 2 mg/l followed by a slow steady increase to values on the order of 6 mg/l at 300 days (Table 1). The columns carrying the metal sludge showed no early peaks of phosphate breakthrough, due most likely to binding with heavy metals and the abundance of calcium in the waste sludge; final levels were similar to those detected in leachate from the control column. Since pH values in the leachates were consistently in the range of 5 to 7, the dominant phosphate species to be expected would be the $H_2PO_4^-$ and HPO_4^{-2}. An examination of literature regarding phosphate complex formation constants indicates that, with the exception of the ferric ion, these species are very weak ligands for any metals likely to be present and of concern in typical leachates. This poor complexing ability combined with the low ($<10^{-3}$ M) concentrations of phosphate in most leachates assures that phosphates will be of relatively little significance in control of metal behavior in landfill leachates.

Ammonia

Except for the control column which displayed a rapid early washout of ammonia, ammonia levels were consistently at the level of ≈0.4 mm M NH_3-N throughout the studies. This low value is rendered even lower by the pH which assures that the effective ligand, NH_3, will be present at only the 0.55 $\mu\underline{M}$ level with the balance being present as the ammonium ion. Countering this is the exceptional ability ammonia has to complex certain of the metals of interest, i.e.,

$$Cu^{+2} + 4NH_3 \rightleftharpoons Cu(NH_3)_4^{+2} \quad \log \beta_4 = 13.0$$
$$Hg^{+2} + 2NH_3 \rightleftharpoons Hg(NH_3)_2^{+2} \quad \log \beta_2 = 17.8$$
$$Hg_2^{+2} + 2NH_3 \rightleftharpoons Hg_2(NH_3)_2^{+2} \quad \log \beta_4 = 17.5$$
$$Zn^{+2} + 4NH_3 \rightleftharpoons Zn(NH_3)_4^{+2} \quad \log \beta_4 = 9.3$$

Considering these systems relative to metal hydroxide solubility in the absence of sulfide demonstrates that only the Hg^{+2} and Hg_2^{+2} will complex to a significant extent with ammonia under the conditions normally present in leachates. For each of these metal species, ammonia complex concentrations in the range of 10 $\mu\underline{M}$ to 10 $m\underline{M}$ can be predicted to exist in equilibrium with the observed ammonia concentrations. This suggests that, in the absence of sulfide, ammonia may have a major impact on the behavior of any mercury which migh be present in landfill leachates. On the other hand, copper and zinc complexes with ammonia will be very minor, approaching maximum concentrations of no more than one micromolar at the highest pH levels.

Carboxylic Acids

Data from volatile acid analyses and the hydroxamic acid test for carboxyl groups suggest that high levels of carboxylic acids in leachates from the subject columns were present (Table 1). Since with the exception of the analysis for volatile acids, identification of the specific species responsible for the carboxyle group levels is unavailable, efforts at drawing conclusions concerning the degree of complexation expected from these compounds are unlikely to yield much information of value. However, low molecular weight monocarboxylic acids such as acetic and propionic acids will generally play little role in heavy metal complextion; the log values of the metal complex formation constants for such metals as copper and cadmium are generally samller than two. Ligands having such small constants will simply be unable to compete significantly with other dissolved substances in leachates. Low molecular weight dicarboxylic acids such as oxalic and malonic acids are much more effective ligands; log values are as high as seven for some metals. While constants of this magnitude are substantial, data regarding the concentrations of these or other such acids in leachates are lacking.

Amino Acids and Peptides

Functional group analyses using the ninhydrin method for amino acids following hydrolysis indicated that the leachates studied had amino acid levels on the order of 5 - 10 $m\underline{M}$. If these values represented the free amino acids, the corresponding level of complexing potential would be substantial since, for such elements as copper and nickel, the logs of the complex formation constants are on the order of 15 or high enough to be significant. However, it is more likely that much of this amino acid was present as polypeptides of varying size. Since the formation constants tend to decrease as the number of amino acid units increases [7,8], it is probable that the log of the formation constant for the actual chemcial entities present might be as low as about 10 for copper and nickel. Even at this level, amino acids and polypeptides could play a significant role in the chemistry of many heavy metals in leachate. This role would again be greatly reduced in the presence of sulfide.

Phenolic Substances

Aromatic hydroxyl compounds, as measured by functional group analysis, are major constituents of leachate. Since many of these phenolic-type compounds have a substantial ability to complex heavy metals they could be of great significance in evaluation and interpretation of heavy metal behavior in leachates. For instance, copper forms strong complexes with low molecular weight phenolic compounds such as salicylic acid and catechol.

$$Cu(\text{Salicylic Acid})_2^{+2} \quad \log \beta_2 = 15.7$$

$$Cu(\text{Catechol})_3^{+2} \quad \log \beta_3 = 41.2$$

Unfortunately, the nature of phenolic substances is not well understood beyond the likelihood that they exist as moderate to high molecular weight substances of the types generally referred to as humic substances. This weakens efforts at drawing conclusions based on smaller compound equilibria. However, it is obvious that the complexing ability of many phenolic compounds for heavy metals is high and that the humic substances will share this complexing capacity. Indeed, some evidence exists [15] that the logs of the formation constants for heavy metal-fulvic acid complexes are on the order of +20. On this basis, it seems probable that the behavior or heavy metals in leachates will also be strongly linked to their interactions with aromatic hydroxyl compounds.

SUMMARY AND CONCLUSIONS

Studies on the removal of heavy metals from landfill leachates have been extended to further consider the importance of ionic strength, chemical activity, complexation, precipitation and filtration when the landfill is operated under conditions of leachate containment and recycle. Examination of leachates obtained from the codisposal of residential-type refuse with metal plating sludge, indicated that ionic strengths of these and many other leachates were sufficiently high so that equilibium - based predictions of reaction potential would need to be corrected for activity. Application of activity coefficients to the reactions describing chemical behavior of important leachate constituents resulted in major modification in such factors as solubility and the availability of sulfides for interaction with heavy metals.

Chemical activity was determined to be responsive to variations in ionic strength and pH of the leachate in a predictable and consistant manner and to reflect the nature of the concomitant biological tranformations occurring in the leachate and landfill mass. Moreover, chemical characterization of the leachate provided strong evidence that chemical complexation is potentially a significant factor influencing the transport and fate of heavy metals. Several inorganic and organic ligands were identified at sufficiently high concentrations to exert a strong influence on the solubility of one or more of the heavy metals.

Finally, the present studies reinforced previous indications that, with leachate recycle and with even very small concentrations of sulfides, most heavy metals will undergo precipitation as sulfides and removal by filtration. Therefore, the use of leachate containment and recycle not only accelerates stabilization processes and the establishment of biologically mediated reducing conditions favorable for sulfide formation, but provides an in situ physical-chemical process for immobilization of heavy metals and reduction of potential for external environmental impairment. Moreover, from the results of these studies, it may be possible for many landfills to receive and detoxify certain types of industrial wastes without causing irreversibe damage to either the stabilization process indigenous to the landfill or to the surrounding environment.

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OPTIONAL COST MODELS FOR SOLID WASTE DISPOSAL

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ABSTRACT

This paper presents our findings from an analysis of 45 landfills, and associated transfer station, balers, shredders, and transportation networks. The analysis of the site attempted to determine how much it costs to build and operate a landfill and which factors have the greatest impacts on those costs. The studied landfills ranged in size from under 100 to over 5,000 tons per day and were located across the continental United States.

A primary concern of the study was to determine the reduction in landfill costs from waste baling or shredding. Analysis of the data indicates that the savings incurred at the landfill, because of prior baling or shredding, does not compensate for the added cost of the more sophisticated processing facilities for the average case. On the other hand, baling or shredding may be feasible in a situation where component costs for the entire landfilling system are exorbitantly high.

Another important finding is that landfilling is only a small portion of total costs. On average, both the haul and the processing components of the system are larger than the landfilling component. The following paper presents the findings from our analysis, the implications of those findings and a methodology designed for use by local communities in determining their most cost-effective landfill system.

How much does it cost to build and operate a sanitary landfill? Which factors have the greatest impacts on cost: size, location, ownership, depth to groundwater, baling or shredding?

Those two questions were the basis for a recent study of landfill costs, performed by Urban Systems Research and Engineering under contract to the Environmental Protection Agency, Municipal Environmental Research Laboratory. The study analyzed costs at 45 landfills, and associated transfer stations, balers, shredders, and transportation networks. The landfills ranged in size from under 100 to over 5,000 tons* per day and were located across the continental U.S. Data on current landfill capital and operating costs has been used to develop estimating procedures, which may be useful in evaluating alternatives for your next landfill site. This article describes the sites, the data collection procedures, the results and how the results can be used.

1.0 The Sample of Landfills

In choosing sites, five basic conditions had to be met on available data:

- the system had to have, and use scales,
- the site had to have been in operation at least one year,

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*"Tons" in this report are metric tons 1000 kg; cost per ton is about 91% of the cost per ton.

- o the operator had to be willing to provide us with both operational and cost data,
- o the bulk of the waste stream had to be residential/commercial, and
- o resource recovery, if any, could only be source separation, and hand-picking.

Since the results were expected to be clearest in dollars per ton, the overriding prerequisite was the existence of scales and weight records. In addition, because of RCRA, only landfill sites which state regulatory agencies considered to have at least acceptable (and preferably outstanding) performand records were selected.

A major concern addressed by the study was any reduction in landfill costs from waste baling or shredding. For that reason, the original research design called for up to one-third of the sites to be balefills and one-third to be shredder waste fills. However, there are only 12 balefill and nine shredded-waste landfills in the country which meet the five conditions listed above. Another objective of the original research plan was to collect a significant amount of data from sites landfilling less than 100 metric tons per day. Primarily because of the lack of scales at small sites, the sample is more representative of medium-scale or larger operations.

After hundreds of phone calls and exhaustive use of various site surveys, 45 acceptable sites were selected. Figure 1 shows their location, and Tables I and II provide a general classification by type and size.

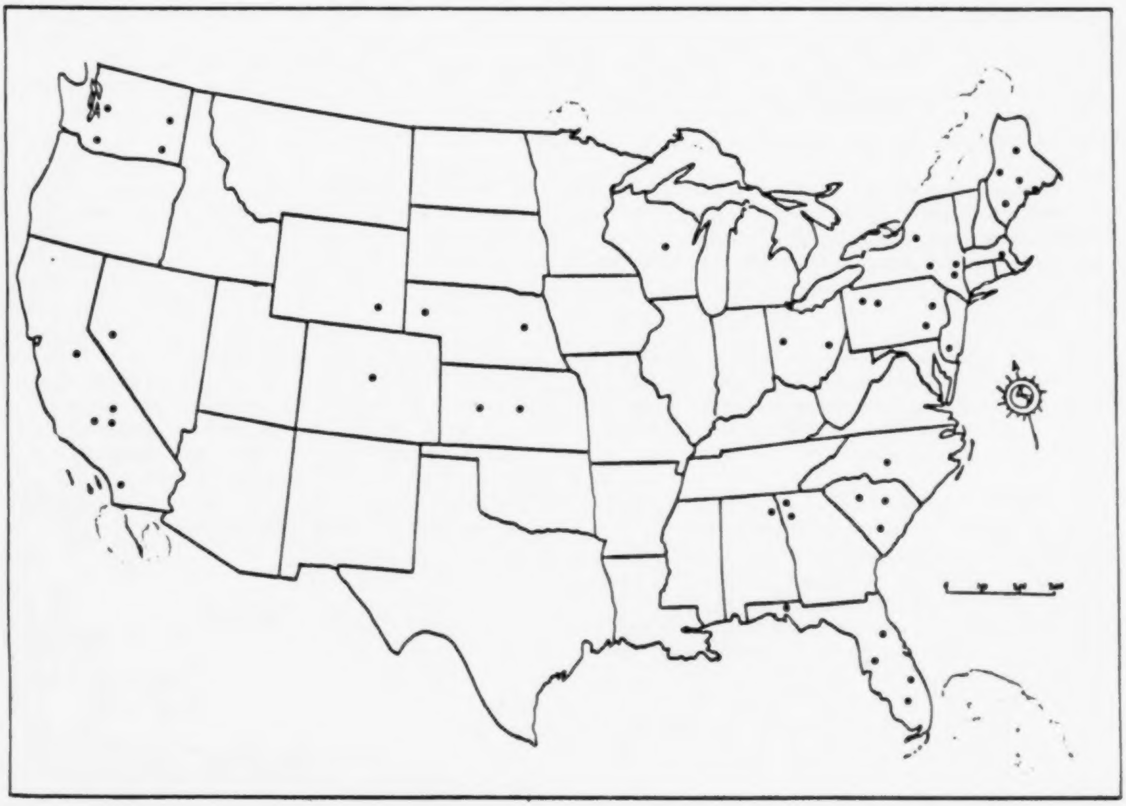


FIGURE 1: **LOCATION OF THE 45 CASE STUDY SITES**

TABLE I. SIZE DISTRIBUTION, BY PROCESSING TYPE, OF CASE STUDIES
(in metric tons per day)

Size Type	0-100	100-300	300-600	600+	Total
Direct Haul to Landfill	4	3	2	3	12
Shredder Processing	0	24	1	4	9
Baler Processing	3	3	4	2	12
Transfer Station	1	1	6	4	12
Total:	8	11	13	13	45

TABLE II. NAMES OF NON-CONFIDENTIAL SITES

Direct Haul to Landfill	Shredder Processing	Baler Processing	Transfer Station
Ft. Lauderdale, FL	Onondega County, NY	Portland, ME	Hamden, ME
Grunderville Landfill Warren, PA	Guilford County, NC	Atlanta, GA	New Gloucester, ME
Spadra Landfill, Los Angeles County, CA	Appleton, WI	Pullman, WA	Columbia, SC
Miramar Landfill San Diego, CA	New Castle, DE	Oyster Bay, NY Smithtown, NY	Mission Canyon Landfill, Los Angeles County, CA
Carson City, NV			Sacramento, CA
Marion, MA			Pensecola, FL
Shawnee County, JN			Lycoming County, PA

2.0 Data Collection and Organization

Data were collected from all sites in January through April, 1980. In an average case, staff from USR&E were on site for 1-2 days, observing the operation and collecting financial and operational data. This was not a detailed engineering study. No time and motion analyses were conducted nor were actual measurements of cell size of dozer horsepower made. Instead, the focus was on equipment and labor allocations and costs. For operating costs, the most recent fiscal year was used; for capital costs, data were collected on the actual equipment in use, age of the equipment, and purchase prices.

In all cases, the data were organized according to the disposal process, as shown in Figure 1. It was hypothesized that, for example, baler costs would depend on capacity, transportation costs on distance, and landfill costs on whether the waste was baled or not. To test these hypotheses, it was necessary to allocate the costs to the various parts of the disposal process. By doing so, the study could differentiate between costs for a baler and costs for the transfer truck that took the bales to the landfill.

The only unusual allocation is the differentiation between "landfill construction" and "landfilling." Landfill sites come in all types,. Site preparation may range from "none" to "two liners, hundreds of acre-feet of excavation, a leachate collection system, and 10 monitoring wells." These landfill construction costs are extremely site-specific. The costs of actually placing and covering the waste on a prepared site, landfilling, are less site specific, except where cover must be purchased off-site.

Obviously, landfill construction and landfilling may interact. Site preparation is often phased, occurring simultaneously with landfilling of waste. Frequently, equipment is used for landfilling on peak days and for landfill construction during off peak times. Still, with the help of the site operators, it proved possible to allocate the costs; this greatly improved the results.

The final topic to be addressed under this section is standardization. To compare costs from site to site, it was necessary to adjust for a wide variation in labor and capital costs. Wage rates varied considerably (see Table III) and fringes and overhead varied even more. Typical values were chosen for each, and standardized labor costs for all sites were calculated using actual hours and typical hourly rates.

It was more difficult to standardize capital costs, because of the diversity in equipment and differences in accounting practices. However, design lives, depreciation schedules, and interest rates were set for fixed and mobile equipment and buildings. We assumed no depreciation on land. For older equipment, either new quotes were obtained for comparable items or the ENR Construction Costs Index was used to convert actual costs to estimsted costs for July 1979 purchase. Standard present worth tables provided the equivalent annual cost used in the analysis. These mean assumptions for standardization are shown in Table IV.

3.0 Descriptive Results

Table V shows the costs for the different steps in the process, along with the relevant sample sizes. The sample sizes indicate the sites for which there were complete data collected for each particular component. In some cases there is complete data for some components while there was insufficient data on another component. For example, it is difficult to estimate landfill construction costs when all the excavation is done using whatever department has free at the time. On the other hand, the landfilling data may be adequate.

Based on Table V, it is obvious that landfilling is usually only a small portion of total costs. Consider a complete hypothetical system: collection vehicles haul to a transfer station, larger vehicles continue to the landfill, and the waste is landfilled there. If all costs are at the average for all sites, the total is $28.95 per metric ton, with landfill accounting for only $6.88. For this average system, a baler would add $17.58 but the transfer station would be replaced; the charge would increase processing costs by $7.83. Even if landfill costs were reduced to zero because of the baler, the systems costs would increase to $29.90. A baler would obviously be a bad choice in this particular situation. On the other hand, if all costs are "maximum" rather than average, a baler would be a good investment.

TABLE III. 45-SITE AVERAGE WAGE RATES (WITHOUT FRINGES)

	Average Over All Sites	Lowest Value	Highest Value	Sample Size
Collection	5.92	3.50	10.62	20
Large-vehicle transport	5.74	3.57	8.87	31
Processing	5.62	2.90*	8.21	28
Landfilling	5.66	3.27	8.82	42
Landfill construction	5.83	4.20	9.33	27

*Below minimum wage.

TABLE IV. ASSUMPTIONS FOR STANDARDIZATION OF LABOR AND CAPITAL COSTS

Labor Assumption:	
Standard hourly wage	$ 5.33
Standard hourly fringe benefits	33%
Capitalization Assumptions:	
Moving equipment	5 years @10%
Stationary equipment	10 years @8%
Buildings	20 years @8%

TABLE V: COST IN DOLLARS PER METRIC TON

	Sample Size	Average	Standard Deviation	Minimum	Maximum
Haul in collection vehicles					
collection	20	1.18	.79	.08	3.41
operating	20	5.67	3.57	.40	15.21
Total:	20	6.85	4.69	.47	18.62
Transfer stations					
capital	10	2.53	4.28	0	14.58
operating	10	7.22	10.60	.33	32.55
Total:	10	7.75	14.23	.83	47.12
Shredder					
capital	7	4.95	2.54	2.00	8.44
operating	8	5.63	2.52	2.31	9.48
Total:	7	11.06	3.75	6.49	17.93
Baler					
capital	11	9.58	5.82	1.45	19.76
operating	11	8.00	4.68	2.31	17.75
Total:	11	17.58	9.50	6.07	31.61
Haul in large vehicles after processing					
capital	20	2.15	3.26	.43	15.29
operating	18	3.30	2.39	.95	10.97
Total:	18	5.48	4.68	1.74	19.03
Landfilling					
capital	37	1.27	.90	.01	3.89
operating	36	3.20	3.47	.40	16.04
Total:	36	4.46	3.89	.84	19.92
Landfill construction					
capital	24	1.34	2.09	.02	7.54
operating	23	1.14	1.95	.02	7.82
Total:	23	2.42	3.94	.07	14.97

The costs presented in Table V are averages. The landfill costs include normal landfills, balefills, and shredded waste landfills, as well as sites with direct haul or transfer. Analysis of costs in a complex situation is aided by the use of statistical methods to develop a series of cost curves. Those analyses have been completed and the results are available in the main report, and are discussed in Section 4.0. Here, we will simply summarize a few of our main findings:

- not surprisingly, the main variable affecting cost of haul in collection vehicles is the amount of time spent in haul on an average day.

- for both haul in the collection vehicles and haul in the larger transfer vehicles, increasing vehicle capacity reduced costs. A 10 percent increase in vehicle size led to a 6-7 percent decrease in costs, across all sites.

- baling and shredding had little effect on transportation costs per ton. The major factors affecting transportation costs--distance, travel time, and total system level of operations--were much more important.

- Shredders and balers were, as expected, more expensive than transfer stations, both in capital and operating costs, correcting for other factors such as size. However, the estimate for shredders is probably low: in general, the shredders were less clean, and apparently less well-maintained, than the other facilities. In addition, one shredder was not included in the models because it had been down 325 days in the preceding year as a result of an explosion. To get a good estimate of shredder costs, it is necessary to look over a very long period. At the mean capacity (400tpd) and 1 shift of actual operation, the model estimated costs of:

  --baler $ 13.03
  --shredder 9.34
  --transfer station 3.39

  Figure 3 shows cost curves.

- large processing facilities did not appear to be much, if any, cheaper per ton than small ones. However, maintenance and hours of equipment operation did have an effect on costs. Increasing the hours of equipment operation by 10 percent reduced costs by 6 percent. Increasing the hours by going from one to 1.5 shifts of equipment operation per day would reduce operating costs per ton by 30 percent. In essence, as long as there is waste to process, it makes the most sense to keep the equipment operating.

- with respect to landfilling, there were several key variables (see Figure 4). Increases in capacity reduced costs considerably, with a 200 ton per day facility 45 percent less expensive per ton than a 50 ton per day facility. Baling and shredding also helped; a balefill was 43 percent less expensive per ton than a conventional landfill, while an uncovered shredded waste fill was 70 percent less expensive to run. The extra processing costs do lead to savings in landfilling. Note, though, that the shredded waste fills also seemed to operate under significantly lower standards, and the costs may underestimate new system costs. Two other factors were important: phasing of construction and the price paid for cover. All else being equal, having to buy off-site cover more than doubled the cost of landfilling. Phasing construction--prearing the next area while filling the current one reduced landfilling costs, probably because it allowed more efficient use of equipment. However, this was not a particularly large or important variable.

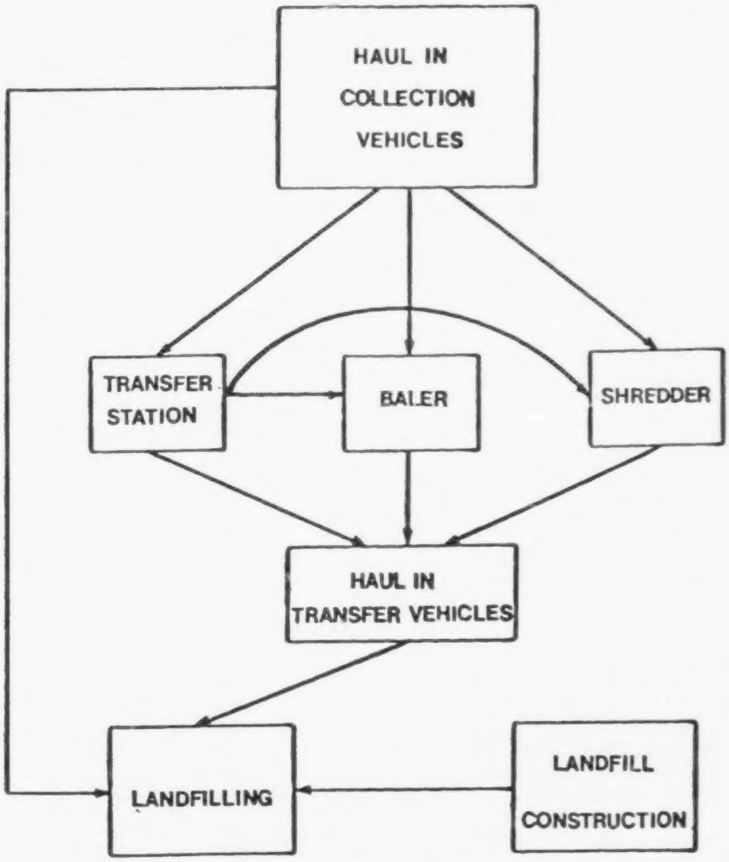

HAUL IN
COLLECTION
VEHICLES
TRANSFER STATION
BALER
SHREDDER
HAUL IN
TRANSFER VEHICLES
LANDFILLING
LANDFILL
CONSTRUCTION

FIGURE 2: Disposal Process

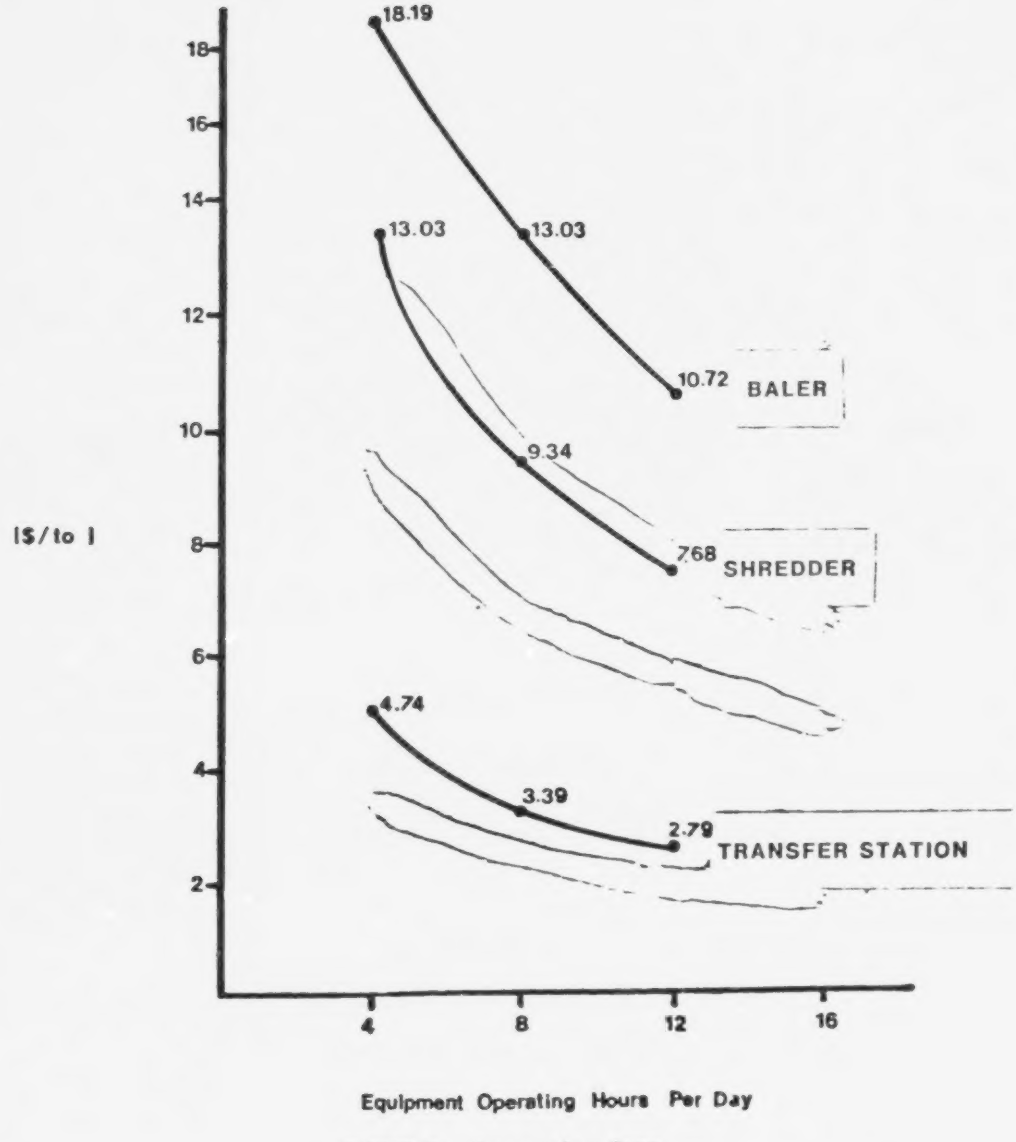
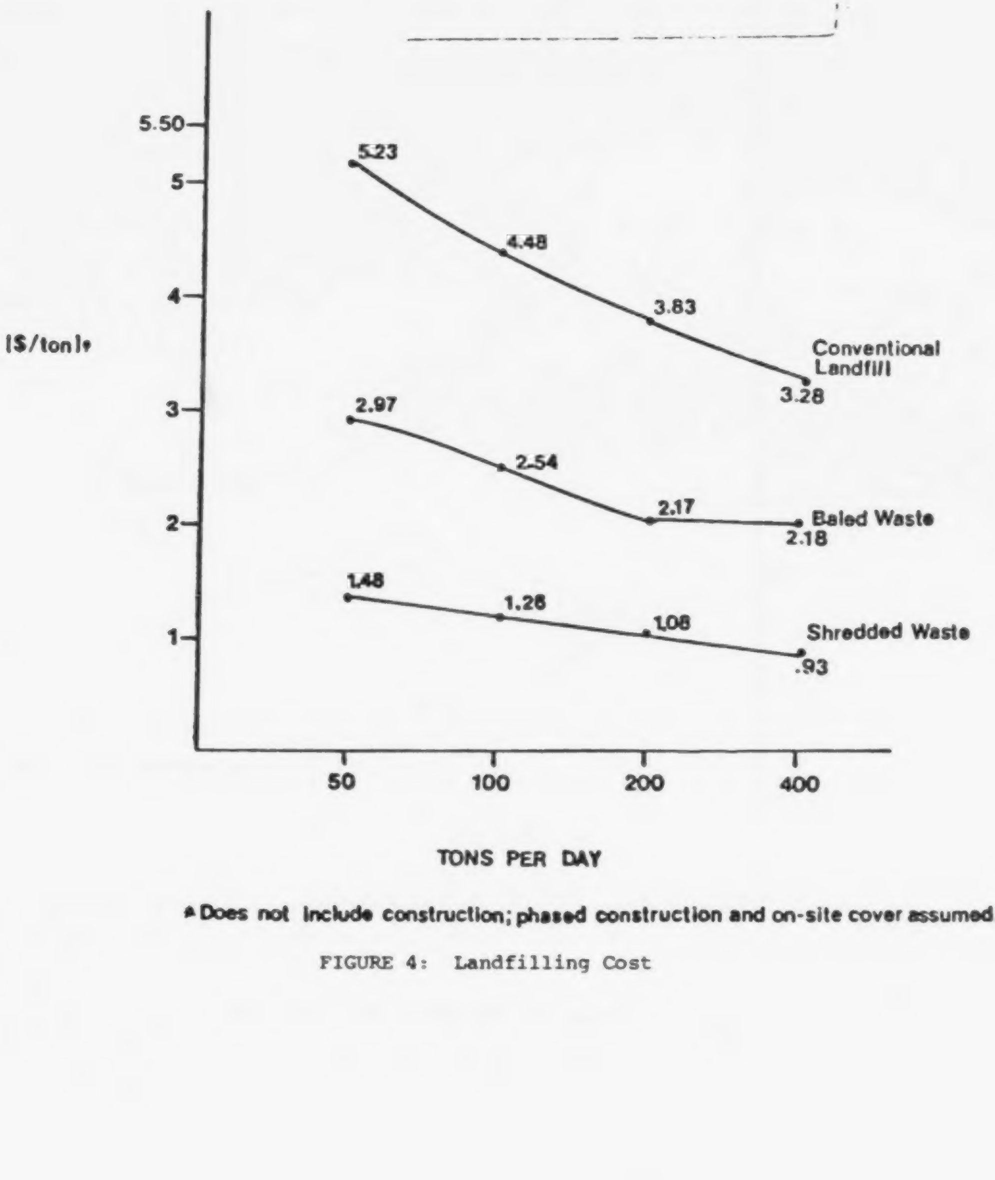


FIGURE 3: Processing Cost



* Does not include construction; phased construction and on-site cover assumed.

FIGURE 4: Landfilling Cost

- we could not model landfill construction. The costs varied too widely and were influenced by a large number of site-specific characteristics. An engineering cost estimate for the site preparation is recommended as the best approach.

Several variables showed no significant effect including:

- ownership (public/private)
- weather (amounts of rain and snow)
- shredder particle size
- bale size

As known, these factors can be important in specific situations. However, as measured at these sites, they seem to be generally less important than the variables discussed above.

Two of the sites deserve special mention, because of their costs. Both were small landfills in an eastern state with stringent groundwater protection regulations. Both required extensive liners, monitoring systems, clay cover, and other special treatment. They were the best examples in our sample of landfills which would meet stringent environmental requirements under RCRA. Our average cost for landfill construction at other sites in the 50-150 ton per day category was $1.34. These two sites had costs of about $14 per ton, though the landfilling costs once construction was completed were about the same as for other sites in that size range.

4.0 Modeling Results

Table VI shows the models. All are in log-log form, so the coefficients can be viewed as elasticities of cost with respect to the other variables. The list of variables is provided at the bottom of the table, and regression statistics are included. The fits are not especially good, but the models do explain a significant portion of the variance. The standard errors for each coefficient are also not small, partly because of the small sample size and measurement problems, but all the major coefficients are significantly different from zero, based on t-test. As planning models, these results should be sufficient and useful.

5.0 Using the Results

Many system operators face a common problem. Their current landfill is being filled, and there is no obvious next site. The options available may include:

- stretching the life of the current site through processing, better compaction, or resource recovery
- making a future site more acceptable through processing
- going further away to find an acceptable site
- entering into regional agreements involving large landfills or resource recovery facilities

Costs are not the only concern in evaluating these options, but they are one of the most important. The toold developed as a result of this study and presented here may be helpful in determining whether a particular option bears further consideration. Although detailed evaluation will require good, site/specific engineering cost estimation, rough comparisons are useful when the number of choices is large and the data are limited.

Consider a hypothetical system: 200 tons per day, direct haul to landfill, 50 km round trip. Now, assume that the system operator has five options:

- make no changes
- increase collection truck size 10 percent
- build a transfer station
- build a baler
- build a shredder

We're interested in the cost/ton for each of the options under reasonable assumptions; these have been computed and are shown in Table VII. Note that both baling and shredding increase processing costs and reduce landfilling costs. We have assumed that a balefill or shredded

TABLE VI: MODELING RESULTS

Type	Model	Statistics
Haul in collection vehicles		
capital:	ln(HCOSTC)=3.46 + 1.43ln(TIMFR) -.708ln (.90) (.33) (.31) (TONLD)	R^2 = .54 F(17,3) = 9.95
operating:	ln(HCOSTO)=5.49 + 1.76ln(TIMFR) -.686ln (.80) (.29) (.28) (TONLD)	R^2 = .67 F(17,3) =17.48
total:	ln(HCOST) =5.56 + 1.66ln(TIMFR) -.688ln (.80) (.29) (.28) (TONLD)	R^2 = .65 F(17,3) =16.11
Transfer in larger vehicles		
capital:	ln(TCOSTS)=7.00+.138ln(TIME)-.835ln(TONS) (1.80) (.21) (.21)	R^2 = .70 F(14,4) =10.9
operating:	ln(TCOSTO)=4.88+.373ln(TIME)-.686ln(TONS) (1.71) (.20) (.20)	R^2 - .52 F(14,4) = 5.2
total:	ln(TCOST) =6.86+.287ln(TIME)=.792ln(TONS) (1.26) (.14) (.15) +.794lu(LOADS) (.21)	R^2 = .77 F(14,4) =15.3
Processing		
capital	ln(PCOSTC)=2.50+1.42 BALER+.874 SHREDDER (1.57) (.36) (.38) +.53ln(EAP) - .53ln(HRS) (.27) (.22)	R^2 = .71 F(19,5) =11.8
operating:	ln(PCOSTO)=3.24+1.20 BALER+1.05 SHREDDER (1.64) (.37) (.39) +.21ln(EAP) - .46ln(HRS) (.28) (.23)	R^2 = .61 F(19,5) = 7.6
total:	ln(PCOST)=3.53+1.35 BALER+ 1.01 SHREDDER (1.39) (.31) (.33) +.34ln(EAP)=.48ln(HRS) (.24) (.20)	R^2 = .73 F(19,5) = 12.6
Landfilling		
capital:	ln(LCOSTC)=1.53-.31 BALER- .41 SHREDDER (.84) (.36) (.59) -.40ln(TONDAY)+.66 COVER+ .89 PHASED (.14) (.70) (.44)	R^2 = .27 F(28,6) = 2.1
operating:	ln(LCOSTO)=2.60-.82 BALER - 1.66 SHREDDER (.48) (.20) (.33) -.19ln(TONDAY)+ 1.18 COVER -.70 PHASED (.08) (.40) (.25)	R^2 = .59 F(28,6) = 8.0
total:	ln(LCOST)=2.88-.57 BALER -1.26 SHREDDER (.42) (.18) (.30) -.22ln(TONDAY)+ 2.00 COVER -.35 PHASED (.07) (.35) (.22)	R^2 = .54 F(28.6) = 6.5

HCOST*	haul cost per ton; suffix: (c) = capital (o) = operating
TIMFR	fraction of vehicle and crew time spent hauling waste
TONLD	haul vehicle capacity, tons per load
TCOST	transfer in large vehicles, cost per ton
TIME	round trip travel time, minutes
TONS	total tons transferred per year
LOADS	transfer vehicle loads per year
PCOST	processing cost, $/ton
BALER	is the waste processed by a baler (0 = no, 1 = yes)
SHREDDER	is the waste processed by a shredder (0 = no, 1 = yes)
CAP	design capacity of processing facility, tons per hour
HRS	actual processing equipment operating hours, last fiscal year
LCOST	landfilling cost, $/ton
TONDAY	average landfill operation, tons per day
COVER	is cover purchased off site? (o = no, 1 = yes)
PHASED	is landfill construction phased with landfilling? (0 = no, 1 = yes)

*NOTE: all models estimated in English units. Standard errors are in ( ) under coefficients

TABLE VII. COST PER TON UNDER VARIOUS OPTIONS FOR A 200 TPD HYPOTHETICAL OPERATION

	DIRECT TO LANDFILLING		WITH PROCESSING AND TRANSFER		
	Current	Increased Vehicle Size	Transfer	Baling	Shredding
Haul in collection vehicles	16.80	15.74	7.35	7.35	7.35
Processing	0	0	3.09	11.85	8.49
Haul in transfer trucks	0	0	4.64	4.64	4.64
Landfilling	4.21	4.21	4.21	2.40	1.19
Landfill construction	2.78	2.78	2.78	2.78	2.78
TOTAL:	23.79	22.73	22.07	29.02	24.45

waste landfill have the same environmental requirements as a conventional landfill, so that landfill construction costs are unchanged. We have also assumed, for the moment, that balefills and shredded waste landfills have the same density as conventional landfills.

Increasing collection vehicle size, within reason, can reduce costs by reducing trips; transfer stations may, in some cases, save money. But, in this simple analysis, installation of a baler or shredder would increase costs a great deal.

The question is then raised, would the increased cost of operating a shredder or baler be offset by more efficient use of the existing landfill. We have referred to landfill construction as "site-specific" and suggested getting estimates for each particular facility. Such estimates can provide a cost/acre for site preparation. To determine construction costs per ton of waste, landfilled density must be estimated.

For example, assume a new site can be prepared with construction costs of $10/ton of conventional waste. If using bales increases in-place density by 25 percent, the construction cost would drop to only $8/ton. In areas where baling or shredding increases total waste per acre--either from better packing or reduced cover--these sophisticated technologies may make sense. The data on in-place densities in our sample was limited, at best. This is the area where more field analysis would be useful.

Now consider the next site to be developed, some years hence. It will undoubtedly be more expensive than the current one. It is likely to be a long distance away, subject to more stringent regulations, and hard to find. If that site is going to be much more expensive, it may make sense to try to stretch the useful life of the current site, through baling or shredding. Present value analysis using the cost curves presented in our report can be used to estimte financial benefits of delayed investment in the new site. However, the political benefits of delay may be even more important.

Based on the data from 45 sites, we believe that transfer stations are often justified, but that baling or shredding before landilling will seldom save money over conventional techniques. The advantages of these more sophisticated waste processing technologies are in extending the useful life of a landfill site and reducing demand for cover material.

We also believe that the techniques developed in the study, and briefly described here, can be useful in investigating options for the solid waste disposal system. To aid that use, our final report has been formated as a handbook with examples and work sheets showing how to adapt an actual situation for comparison with our results.

Acknowledgements

This work was performed under EPA Contract #68-03-2868 for the Municipal Environmental Research Laboratory (Cincinnati, Ohio). Oscar Albrecht was the project officer. The final report, entitled "Optional Cost Models for Land Disposal of Municipal Solid Waste" will be available through the National Technical Information Service. Obviously the results are the authors, and do not necessarily reflect the views and policies of EPA.

SUMMARY OF LANDFILL RESEARCH

BOONE COUNTY FIELD SITE

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ABSTRACT

Five municipal waste test cells were constructed at the Boone County Field Site during 1971 and 1972. Two were field-scale, and three were small-scale cells constructed for the purpose of performance comparison. All cells were monitored for temperature, gas composition, settlement, and leachate quantity and characteristics until closure in August, 1980. This paper presents preliminary results from the analysis of over eight years of the leachate quantity and quality data. It must be emphasized that the results are from relatively shallow (less than 2.5 meters) batch-type cells.

INTRODUCTION

The Boone County Field Site consisted of a ten acre tract located 8 km west of the City of Walton, Kentucky. Available facilities at the site included an office trailer, a pole barn, a truck scale, and a weather station.

Elevations at the research site range between 213 and 244 m above sea level. Surficial soils at the site are predominantly a lean clay, classified by USDA as Nicholson silt loam. Rubbly limestone mixed with thin beds of soft calcareous shales of the Fairview formation underlie the shallow soil. The mean annual precipitation in the area is 927 mm. Monthly normal mean temperatures range from 0 deg. C in January to 24.4 deg. C. in July.

The objectives of test cells 1, 2A, 2B, 2C, and 2D included:

(1) Analyze the amount and characteristics of leachate.

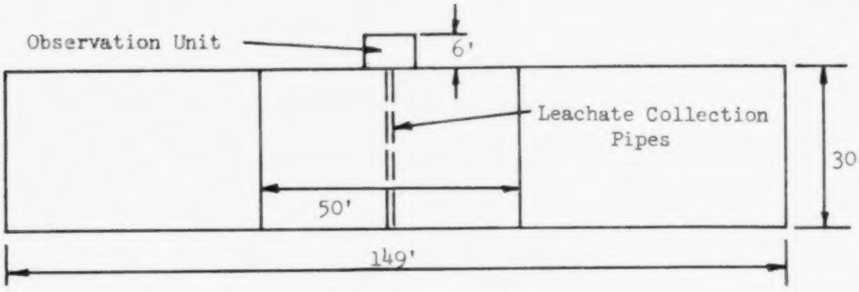
(2) Analyze the composition of gases present in the cells.

(3) Evaluate a clayey silt soil as an impervious liner for leachate control.

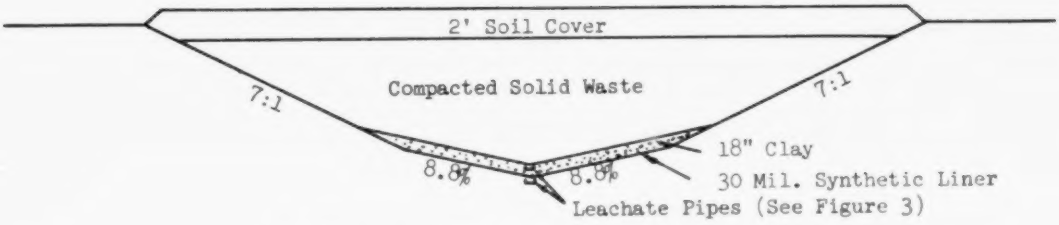
(4) Evaluate the behavior of a field scale test cell, 2D, as compared to similarly constructed small-scale test cells, 2A, 2B, and 2C.

TEST CELL DESCRIPTION

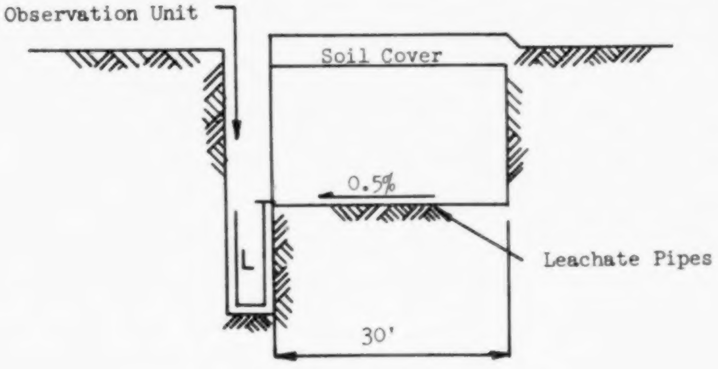
Test Cell 1 consisted of a 45.4 m long by 9.2 m wide trench type sanitary landfill cell. The trench was excavated with vertical side walls and ramps on both ends sloping approximately 1:7. The center 15.3 m of the trench was sloped approx. 7 percent to the transverse center line as shown in Figure 1. A 30 mil (.76 mm) synthetic liner (Hypalon) was centered in the base of the cell to prevent any leachate from migrating below the cell. Directly above the liner a slotted collection pipe


Observation Unit
6'
Leachate Collection Pipes
30'
50'
149'

PLAN VIEW


2' Soil Cover
Compacted Solid Waste
7:1
7:1
18" Clay
8.8%
8.8%
30 Mil. Synthetic Liner
Leachate Pipes (See Figure 3)

LONGITUDINAL SECTION


Observation Unit
Soil Cover
0.5%
L
Leachate Pipes
30'

TRANSVERSE SECTION

Figure 1. Design of cell no. 1.

was installed along the transverse center line. Above this pipe and the synthetic liner a 45.7 cm thick clayey silt soil liner was placed.

A second slotted collection pipe was installed in a trench in the soil liner. Short-circuiting to the lower pipe was prevented by lining the base and sides of the trench with a polyethylene strip. Both pipes drained into an observation unit beside the cell. The area around both pipes was backfilled with clean silica gravel. Details of the drainage scheme are shown in Figure 2.

During June 1971, 395 metric tons or residential refuse was placed in the test cell at a dry density of 429 kg/cubic meter. Moisture, temperature, gas and settlement probes were placed within the cell as it was constructed. Approximately .6 m of the compacted clayey silt (CL) soil was placed over the refuse as final cover.

Four additional test cells containing municipal solid waste were constructed during August 1972. Three of the cells, 2A, 2B, and 2C were enclosed in identical cylindrical steel pipes, 1.83 m in diameter and 3.66 m long. The fourth test cell, 2D, was an 8.53 m square field-scale cell constructed to compare performance with the small-scale cells.

The steel pipes were coated with cold tar epoxy and placed vertically in an excavation on reinforced concrete pads. Slotted pipe in a trough in the concrete pad was used to collect leachate. Earth backfill was placed around the pipes to within 150 mm of the tops. Refuse was placed in the pipes in 90-135 kg increments and compacted by dropping a 135 kg weight from a height of approximately 1.2 m above the refuse. Temperature and gas probes were placed at several levels within the cell during refuse placement. 300 mm of compacted soil was placed above the refuse. Then 300 mm of pea gravel was placed on the soil to allow rapid percolation of rainfall and to minimize evaporation.

Cell 2D was constructed in an excavation 8.53 m square and 3.20 m deep. A 30 mil (.76 mm) CPE liner was placed above a shaped sand bed in the base of the cell and extending up the sidewalls. A slotted PVC pipe was placed along the center line of the base of the cell for leachate collection and gravity drainage to the collection well. The entire base of the cell and liner was covered with 300 mm of silica sand. Plywood sheets were placed against the synthetic liner on the sidewalls for protection from puncture and tearing during cell filling.

A 7 tonne bulldozer was lowered into the cell by a crane for compaction of refuse. Temperature and gas probes were placed during the filling at locations shown in Figure 3. A 300 mm layer of compacted soil cover was placed over the 2.44 m of refuse. A berm system, consisting of 150 mm high triangular-shaped clay berms, was hand constructed on top of the soil cover to promote uniform percolation of rainfall into the refuse cell.

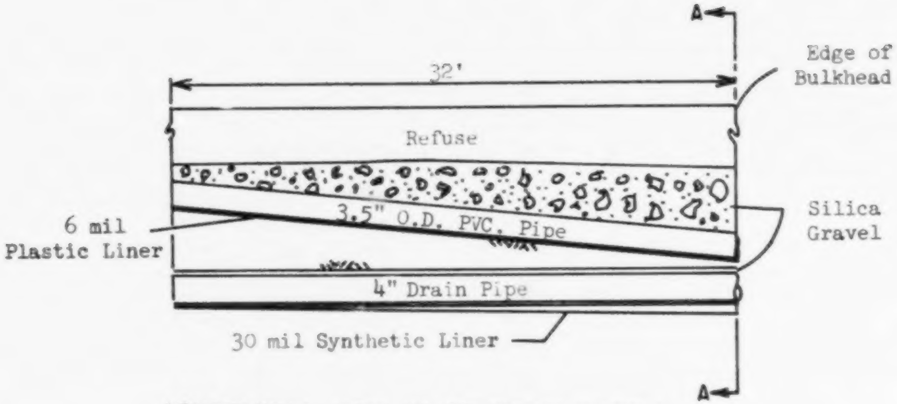
Samples were obtained from the residential refuse being placed in all cells for moisture and composition studies. Chemical analysis of the refuse was done for a number of parameters for Test Cell 1. Statistical analysis indicated that any performance variations within cells 2A, 2B and 2C could not be attributed to the differences in the composition or the original moisture. A summary of the cell data is shown in Table 1.

LEACHATE QUANTITY

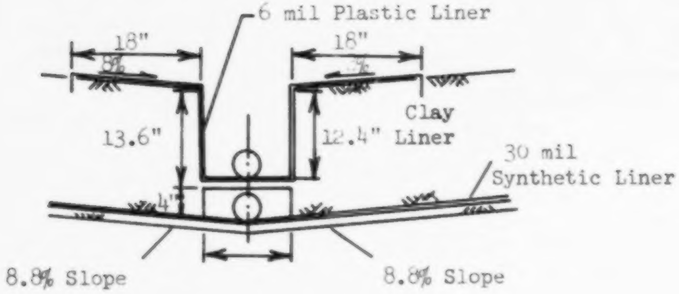
Test Cell 1

Leachate was initially collected from both the upper and lower pipes in Test Cell 1 approximately two months after construction. Cumulative leachate volume is shown in Figure 3 together with the volume of leachate predicted by using the water balance method. Leachate collected was 27% of the precipitation after 6.5 years.

In computing the water balance it was assumed that the monthly mean temperatures for the Boone County site were the same as that reported for Covington, Kentucky.The runoff coefficients for the wet and dry seasons were chosen as .17 and .13. Assuming a soil moisture storage capacity of 125 mm for the final cover, 90 mm of water were required to bring the final soil cover to field capacity. An additional 106,000 liters of water were required to


A
32'
Edge of Bulkhead
Refuse
Silica Gravel
6 mil Plastic Liner
3.5" O.D. PVC. Pipe
4" Drain Pipe
30 mil Synthetic Liner
A

LONGITUDINAL VIEW OF COLLECTION SYSTEM


6 mil Plastic Liner
18"
18"
13.6"
12.4"
Clay Liner
30 mil Synthetic Liner
4"
8.8% Slope
8.8% Slope

SECTION A-A

Figure 2. Leachate collection system.

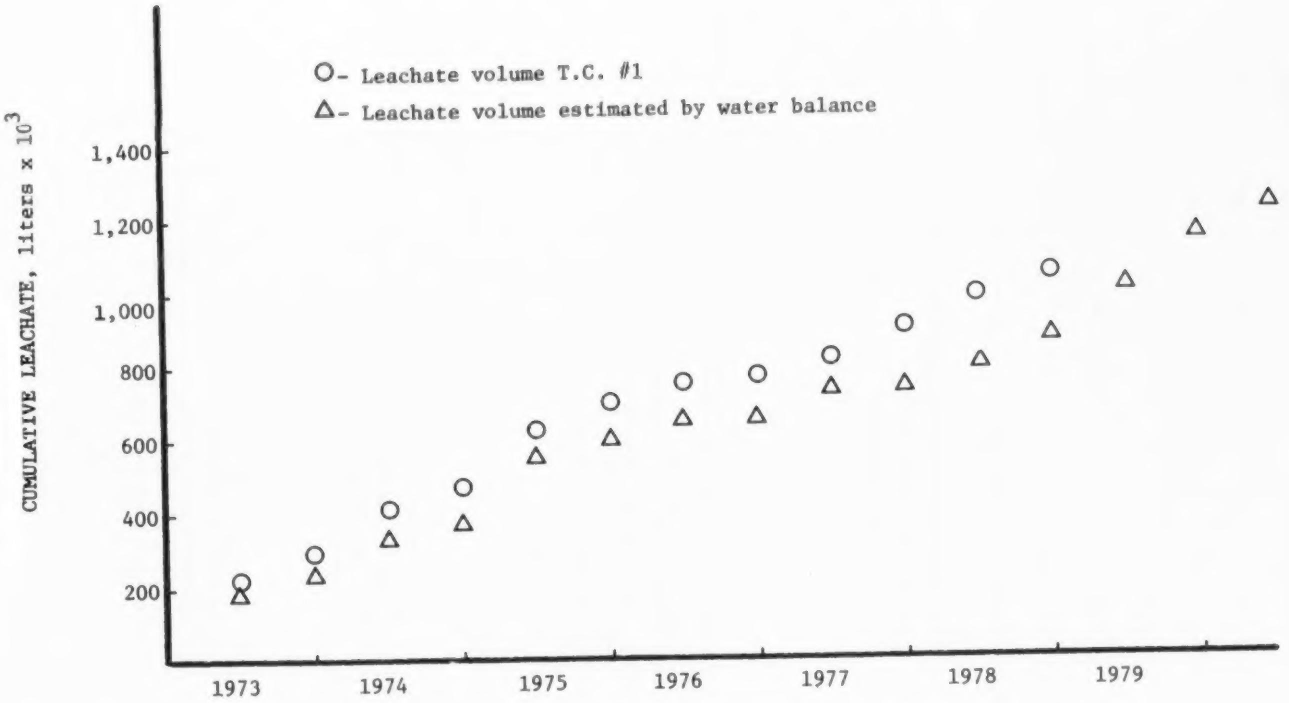


Figure 3. Cumulative leachate volume

TABLE 1. SUMMARY OF CELL DATA

	Test Cell				
	1	2A	2B	2C	2D
Cover soil classification	CL	CL	CL	CL	CL
Depth of soil cover, m	.60	.30	.30	.30	.30
Surface area of refuse, m^2	432.3	2.627	2.627	2.627	72.83
Maximum depth of refuse, m	2.56	2.56	2.56	2.56	2.44
Mass of refuse, kg (dry)	286,000	2,046	2,113	2,135	72,450
Dry density of refuse, kg/m^3	429	304.3	314.1	317.6	407.7
Moisture content of refuse, % wet wt.	27.6	22.5	27.1	24.1	31.8

bring the entire cell and the soil liner to field capacity based on an estimated refuse field capacity of 330 mm/m of refuse.

The water balance calculations were reasonably accurate in predicting the quantity of leachate, with only 16% difference after 6.5 years. If average evapotranspiration values had been used, rather than ones computed from the actual climatic conditions experienced, the difference would have been 43% leachate predicted than actually collected. This large difference indicates a need for leachate volume calculations in designs to be based on extreme weather conditions as well as average values.

Soil Liner

The quantity of leachate collected from both the upper and lower pipes is listed in Table 2. The quantity of leachate from the lower pipe was equal to or greater than that volume from the upper pipe until January, 1972. This was caused by leaving the valve closed on the upper pipe except for weekly sampling, thereby inducing sufficient head to cause leakage into the lower pipe. After December 20, 1971, the upper pipe was allowed to drain freely through an S-trap into a sump in the observation unit, thereby greatly reducing the hydraulic head and consequently reducing the flow from the lower pipe.

The percentage for the leachate collected from the lower pipe decreased over the years due to the change in collection method, and/or a decrease in permeability of the soil liner, to a point where more than 99% of the leachate was collected from the upper pipe.

From Table 2, a tendency can be noted that a greater percentage of the total flow comes from the lower pipe during the months of low total flow. For example, during the period of December 1973-November 1974, 46% of the flow in the lower pipe occurred during the six months when only 31% of the total flow occurred. Apparently during periods of high total flow, the soil liner is saturated, and with greater head existing at the refuse-soil interface the leachate flows principally to the upper pipe.

Soil liner-drain efficiency calculations (5) predict greater than 90% of the leachate should pass through the soil liner. This indicates that the synthetic liner and the soil liner are functioning together and that the larger percentage of the leachate collected in the upper pipe is not due to the soil liner alone. The synthetic liner blocks any deep percolation forcing flow along the refuse-soil interface and resulting in the high percentage of leachate collected in the upper pipe.

TABLE 2. SEASONAL DISTRIBUTION OF COLLECTED LEACHATE

Months	Year	Upper & Lower Pipe Volume - Liters	Lower Pipe Volume-Liters	% Through Lower Pipe
June-Aug.	1971	223	107	48
Sept.-Nov.	1971	376	243	65
Dec.-Feb.	1971-1972*	2789	539	19
Mar.-May	1972	7944	661	8
June-Aug.	1972	1552	66	4
Sept.-Nov.	1972	19,608	880	4
Dec.-Feb.	1972-1973	54,964	395	0.7
Mar.-May	1973	95,157	121	0.1
June-Aug.	1973	53,802	178	0.3
Sept.-Nov.	1973	24,835	97	0.4
Dec.-Feb.	1973-1974	81,321	50	0.1
Mar.-May	1974	61,336	131	0.2
June-Aug.	1974	22,565	72	0.3
Sept.-Nov.	1974	41,035	80	0.2
Dec.-Feb.	1974-1975	61,869	51	0.1
Mar.-May	1975	96,325	67	0.1
June-Aug.	1975	16,378	106	0.6
Sept.-Nov.	1975	31,063	81	0.2
Totals		673,152	3925	.58%

* Beginning of freely drained condition on January 1, 1972

Test Cells 2A, 2B, 2C, 2D

The experimental design called for the input of approximately 500 mm of precipitation each year into all of the cells. Average annual rainfall at the site is in excess of 900 mm so the cells were periodically covered, the cylinders with caps and 2D with nylon reinforced Hypalon. After June, 1979 the test cells were left uncovered all the time.

Leachate was initially collected from each test cell on the date and at the cumulative rainfall quantities in Table 3. The early appearance of leachate from 2D, indicative of some short-circuiting, also occurred in Cell 1.

TABLE 3. INITIAL COLLECTION OF LEACHATE

Test Cell	Date Leachate Collected	Cumulative Precipitation(mm)
2A	6-15-73	724
2B	2-13-73	588
2C	6-19-73	765
2D	9-25-72	51

Figure 4 shows the quantity of leachate collected from each test cell with precipitation. Test Cell 2C produced very little leachate during the reporting period in comparison to 2A and 2B. A test boring in the cell did not show any free water stored in the cylinder. It is possible that a leak developed at a welded joint near the surface of the soil cover and very little of the precipitation actually entered the refuse mass.

The quantities of leachate collected from 2A and 2B varied only slightly. By the end of 1979, leachate collected from 2D was twice that per unit of surface area from 2A or 2B.

LEACHATE QUALITY

Leachate samples were initially analyzed on a bi-weekly schedule. Monthly samples were obtained in the later phases. Concentration data was reduced to weighted mean concentrations at the approximate time at which equal intervals of leachate flow were recorded. This normalized the data for each cell so that concentration histories were based on a parameter accounting for the varying leachate quantities collected from each cell. Concentration and mass removal histories normalized by dry refuse weight are presented in Figures 5-16.

A summary of peak concentrations and the leachate volume on the date of the sample is presented in Table 4 for selected parameters. It is notable that all but a few of the peak concentrations for Test Cell 2A occurred within a short time span starting with the onset of leachate production. This was the time period during which or shortly after field capacity was achieved. Apparently this peak concentration period results from the initial water contact with the refuse when the supply of the leachable substances and the contact time are high.

For Test Cell 2B the time span for peaks was somewhat longer than 2A, ranging over a longer time period and greater volume of leachate collected. Test Cell 2B did begin leachate production 4 months earlier than 2A and before field capacity was reached so this range might actually have coincided closely with 2A.

For the field-scale cells, 1 and 2D, the same general trend of most peak concentrations occuring during a certain time span is noted. It does not appear though that the peak concentrations for 2D occurred as early as for 2A, 2B and Test Cell 1. If peaks did occur earlier, then the high concentrations in the leachate must have been either reduced by significant diluting leakage from the sides of the cell or by channeling through the cell. The latter situation would result in field capacity not actually having been achieved until somewhat after estimated water requirements had been met, possibly during the time and leachate volume range when peak concentrations were recorded. Dilution was indicated by the lower magnitude of almost all of the peak concentrations of 2D as compared to 2A and 2B.

Leachate Composition Comparability

One of the primary objectives of the later test cells was to evaluate the behavior of a field-scale test cell, 2D, compared to similarly constructed small-scale test cells 2A, 2B and 2C. The concept was to determine whether similarity existed between individual small-scale cells as well as similarity between the

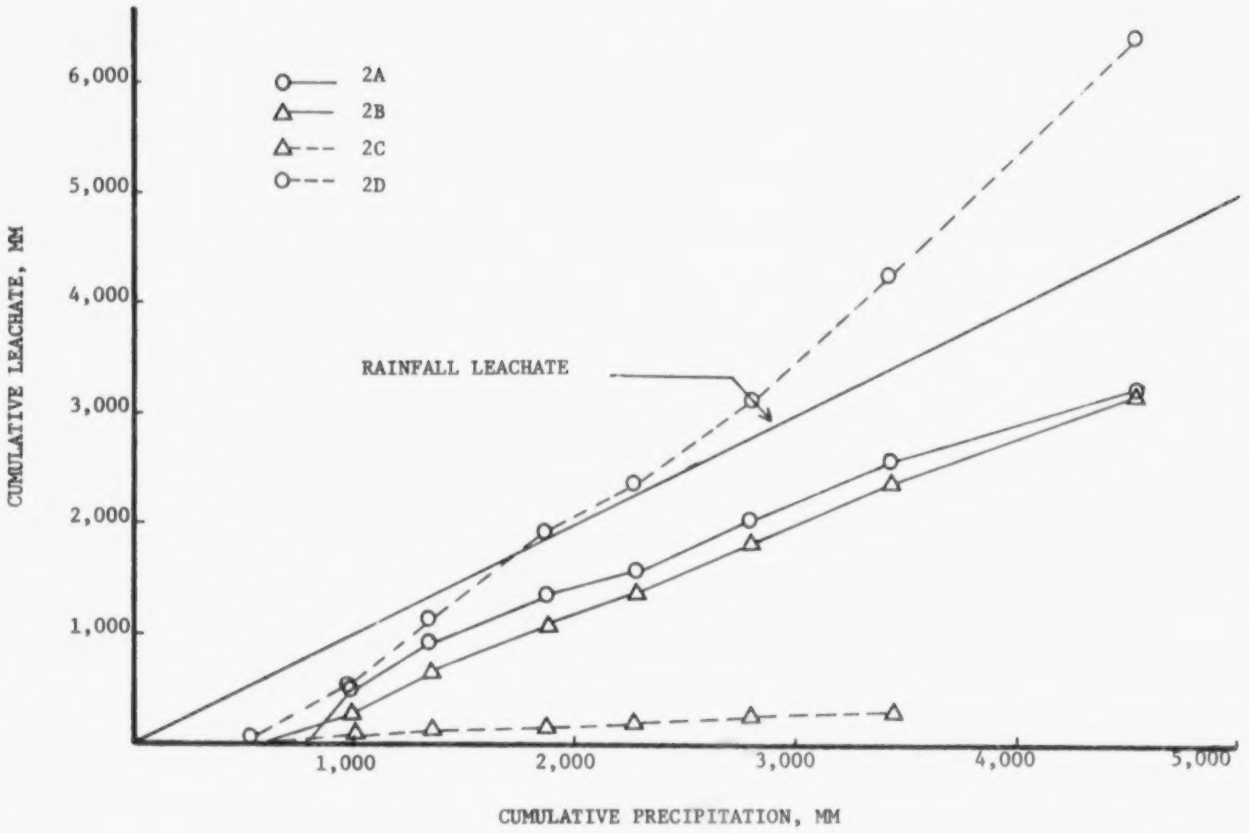


Figure 4. Leachate volume, cells 2A, 2B, 2C, 2D

TABLE 4. PEAK CONCENTRATIONS

	Test Cell 1		Test Cell 2D		Test Cell 2A		Test Cell 2B	
Parameter	Conc.*	Lea. Vol.#	Conc.*	Lea. Vol.#	Conc.*	Lea. Vol.#	Conc.*	Lea. Vol.#
pH	7.07	4.1	7.0		6.2	0.2	6.0	0.3
COD	37,500	0.3	41,869	0.5	57,330	0.2	61,600	0.04
BOD	30,000	2.1	79,120	0.9	62,560	1.0	72,220	0.7
Kjeldahl-N	700	0.4	1,242	0.5	1,560	0.2	1,897	0.3
Ammonia-N	552	0.3	947	0.5	1,035	0.5	1,185	0.3
Orthophosphate	61	0.04	82	0.4	390	0.02	185	0.1
Sulfate	1,160	0.3	1,280	0.5	2,215	2.2	2,775	1.9
Alkalinity	8,870	0.3	8,963	0.7	11,535	0.2	13,880	0.01
Acidity	3,620	0.3	5,057	0.7	6,720	0.06	6,843	0.2
Conductivity	12,200	0.2	16,000	1.0	17,000	0.2	18,000	0.2
Total Solids	23,600	0.3	36,252	0.4	46,484	0.2	45,628	0.2
Sodium	1,049	0.4	1,375	0.4	1,900	0.2	1,700	0.2
Potassium	1,950	0.7	1,893	0.5	2,225	0.1	2,939	0.3
Chloride	1,749	0.6	2,260	0.5	2,335	0.2	2,343	0.2
Iron	614	1.5	1,492	2.4	1,547	0.2	2,902	0.04
Magnesium	374	0.2	411	0.5	486	0.2	617	0.3
Manganese	184	0.04	58	0.4	109	0.02	115	0.04
Calcium	2,320	0.2	2,300	0.6	2,280	0.1	4,000	0.04
Zinc	104	0.2	67	0.3	150	0.1	360	0.1
Hardness	7,500	0.3	6,713	1.9	7,067	0.06	10,575	0.04

*. Concentration in mg/l except for pH and conductivity (micromhos/cm)

#. Leachate volume - liters per dry kilogram of refuse

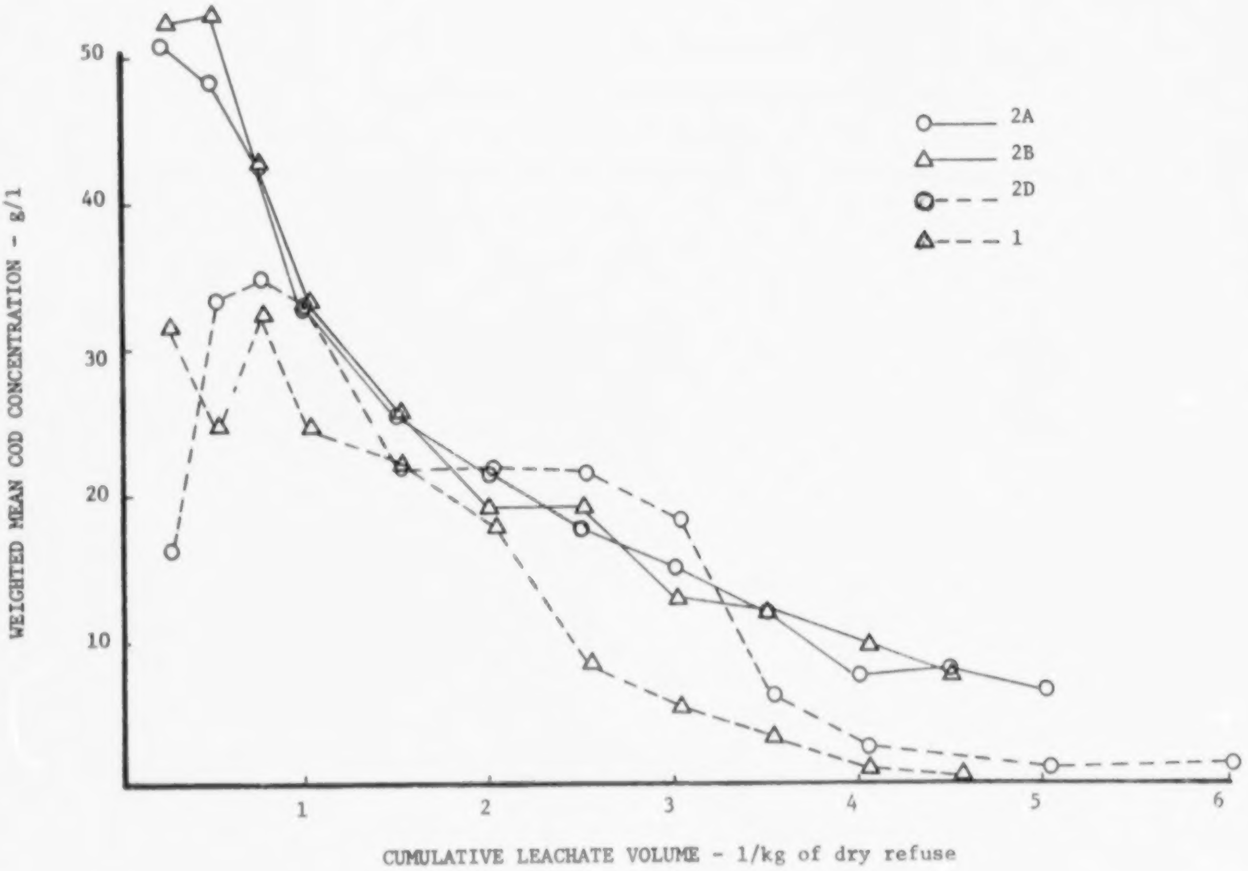


Figure 5. COD concentration history

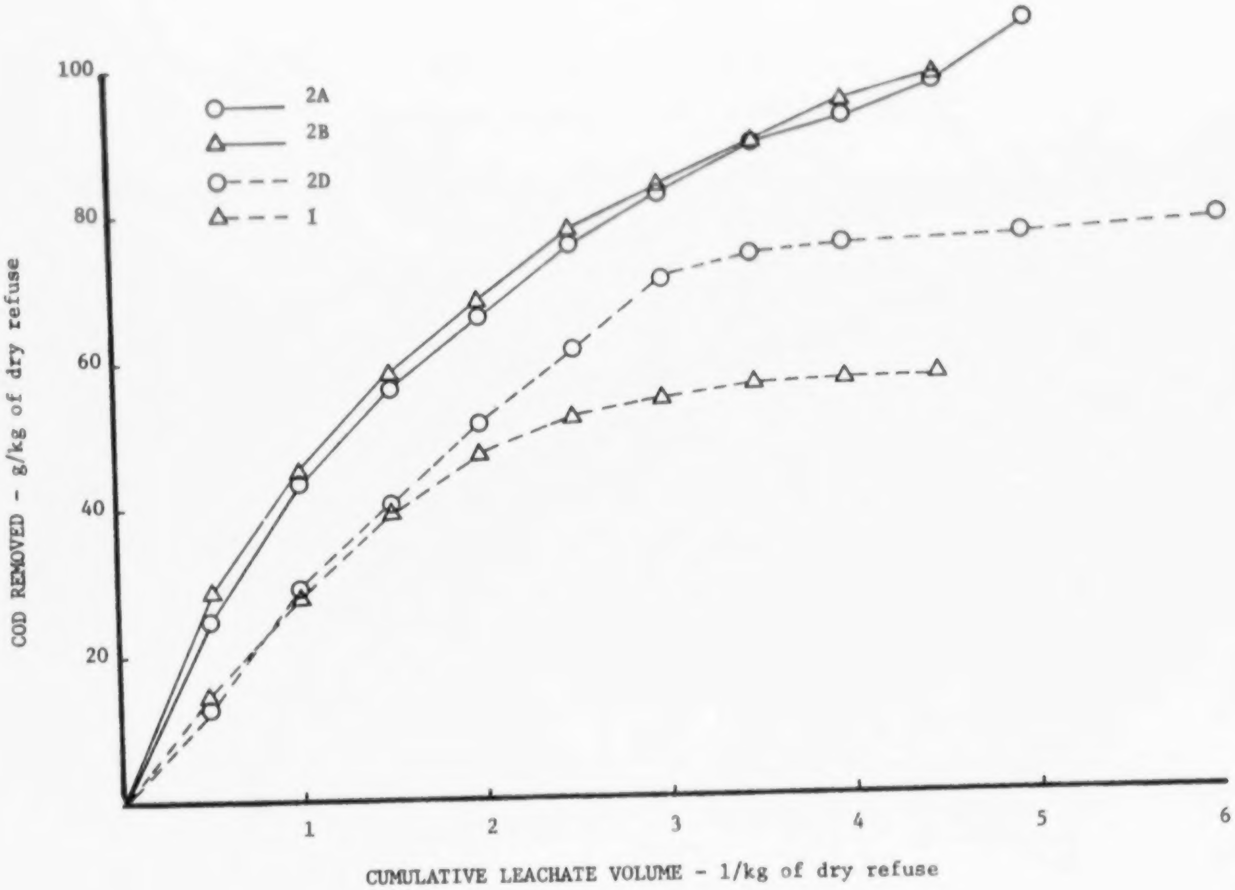


Figure 6. COD mass removal

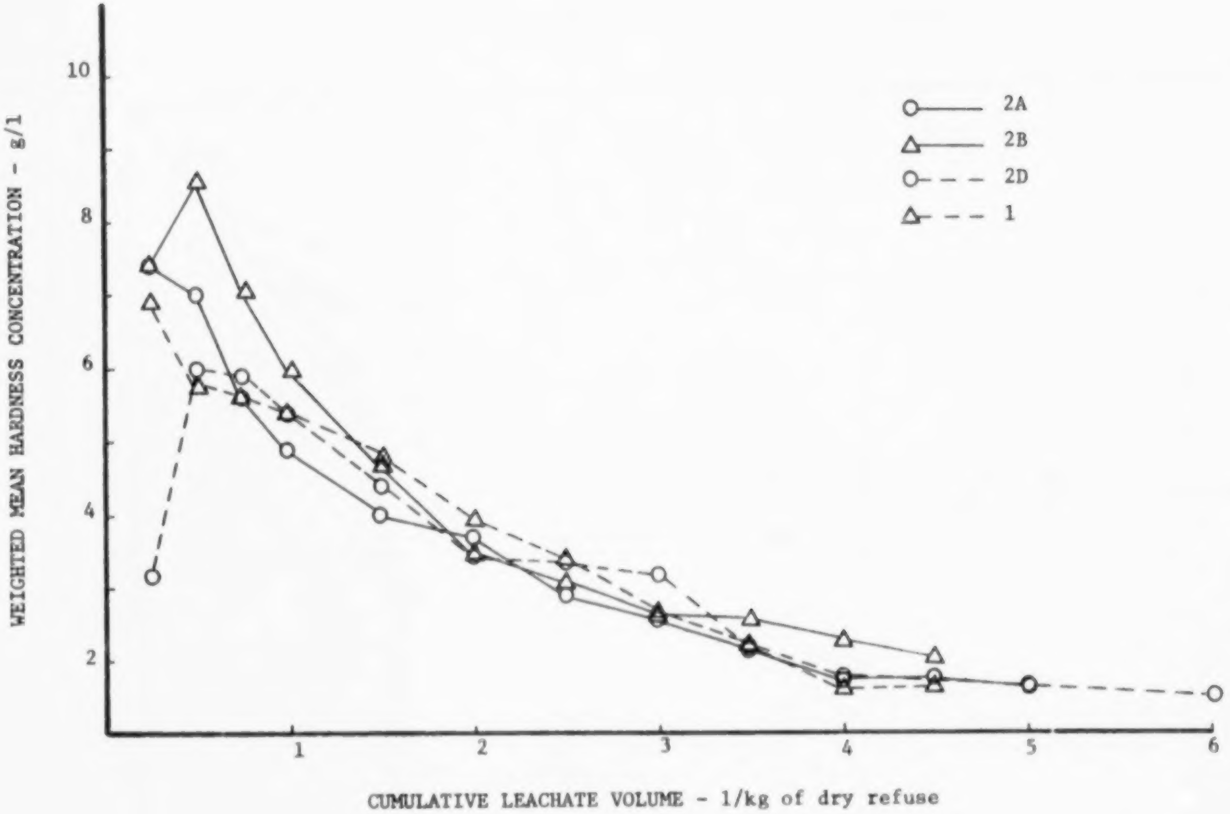


Figure 7. Hardness concentration history

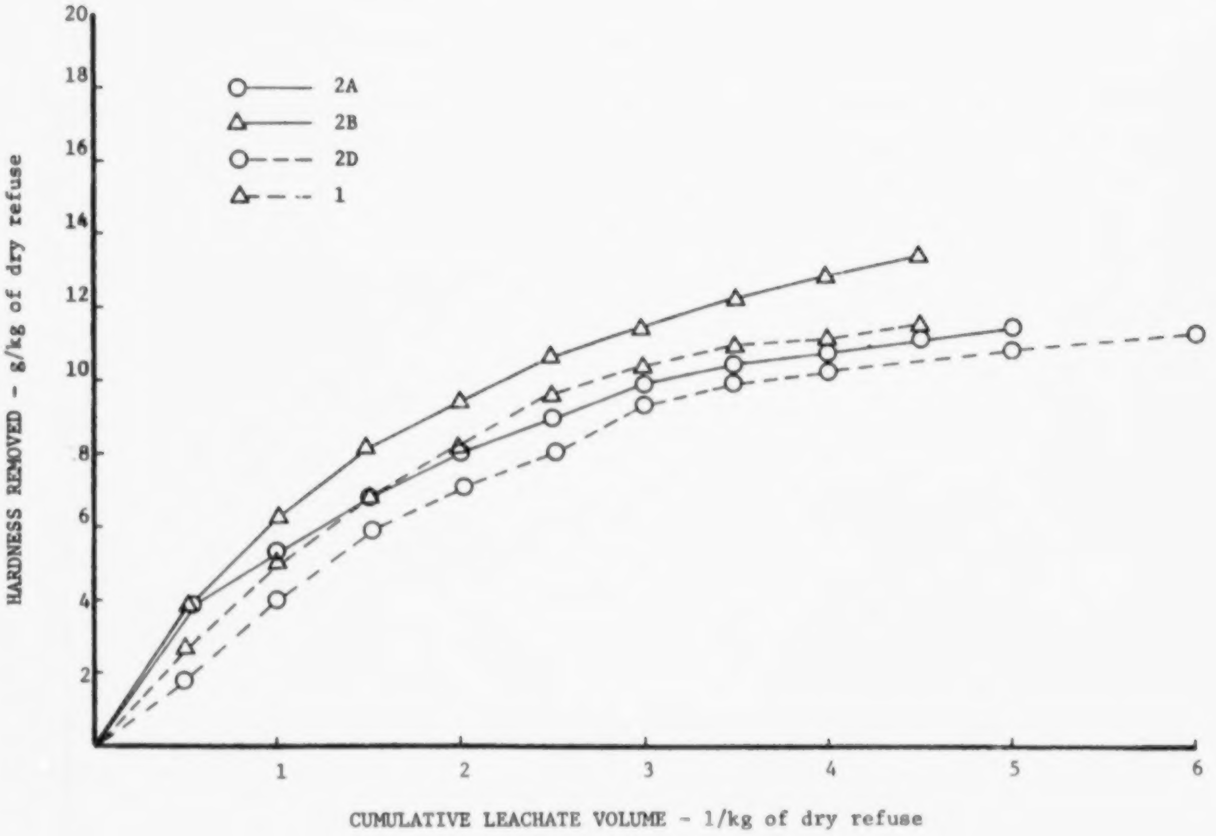


Figure 8. Hardness mass removal

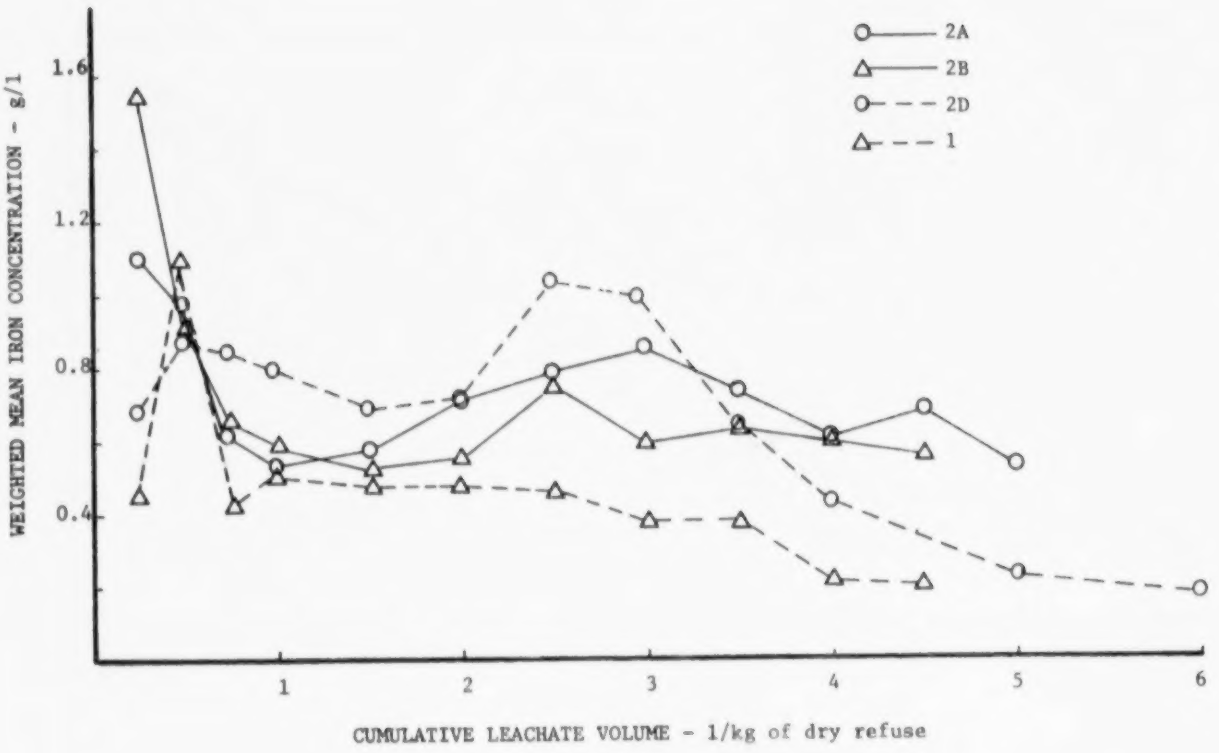


Figure 9. Iron concentration history

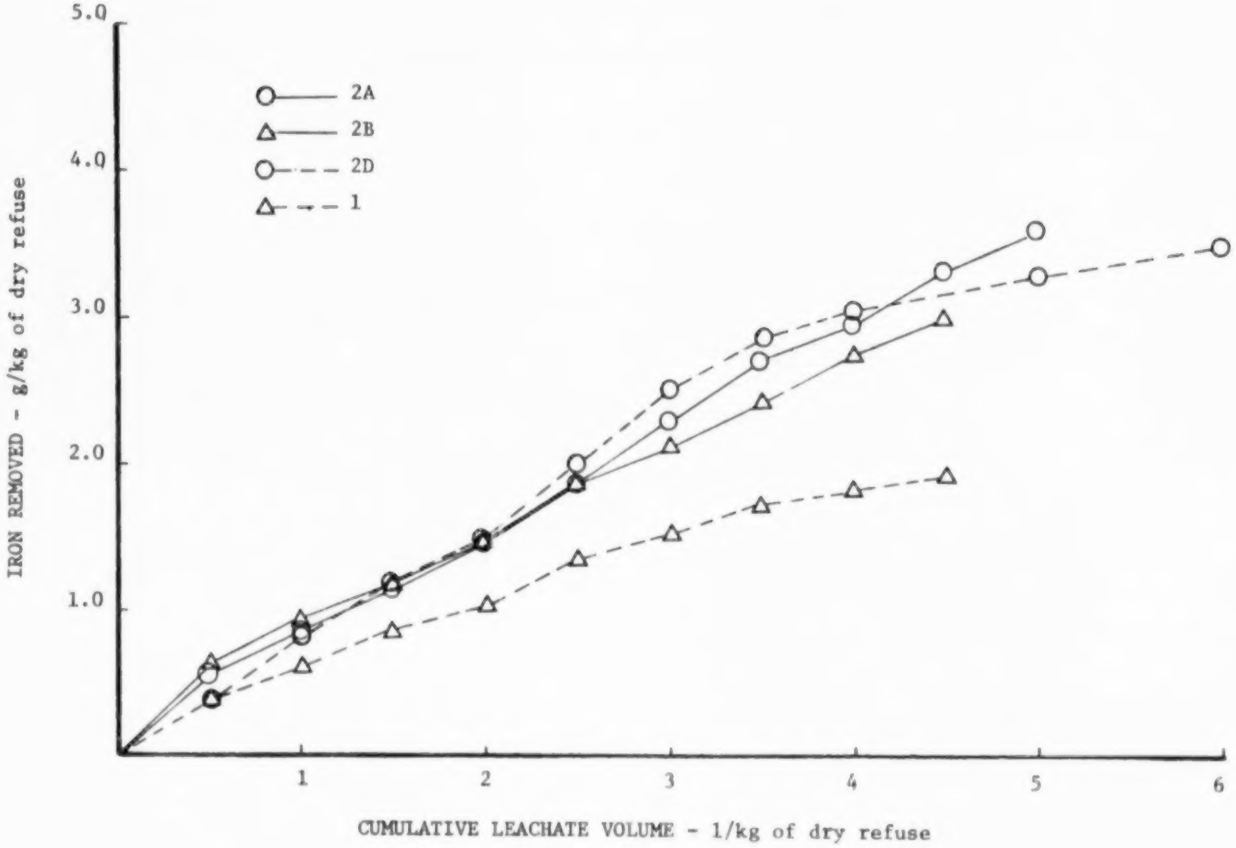


Figure 10. Iron mass removal

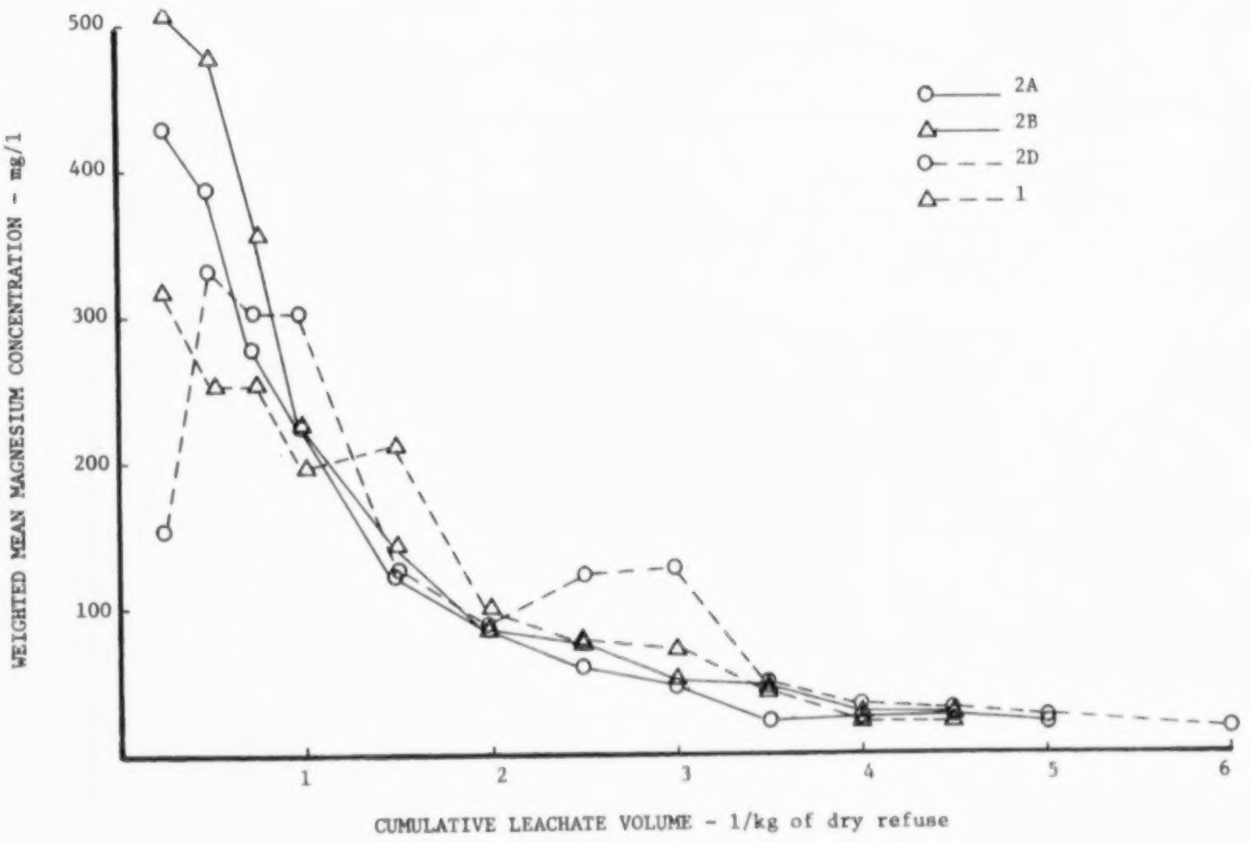


Figure 11. Magnesium concentration history

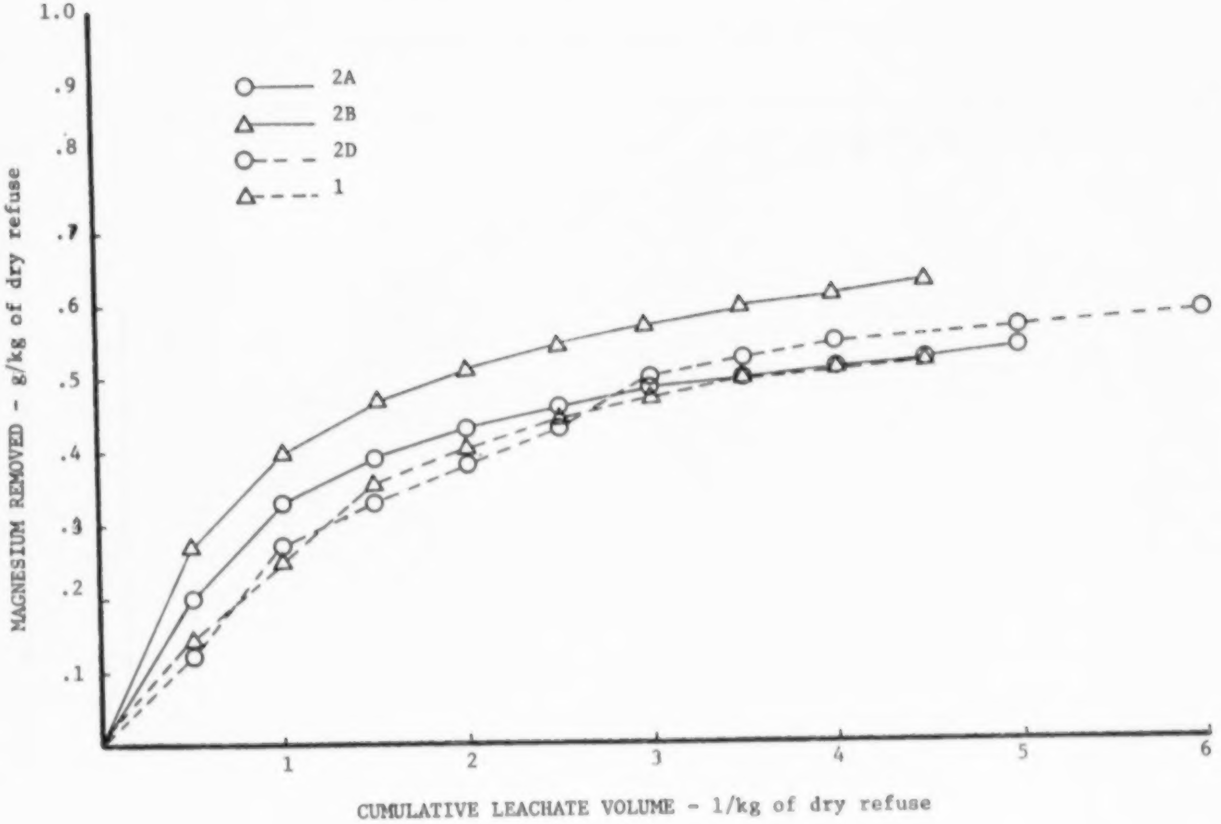


Figure 12. Magnesium mass removal

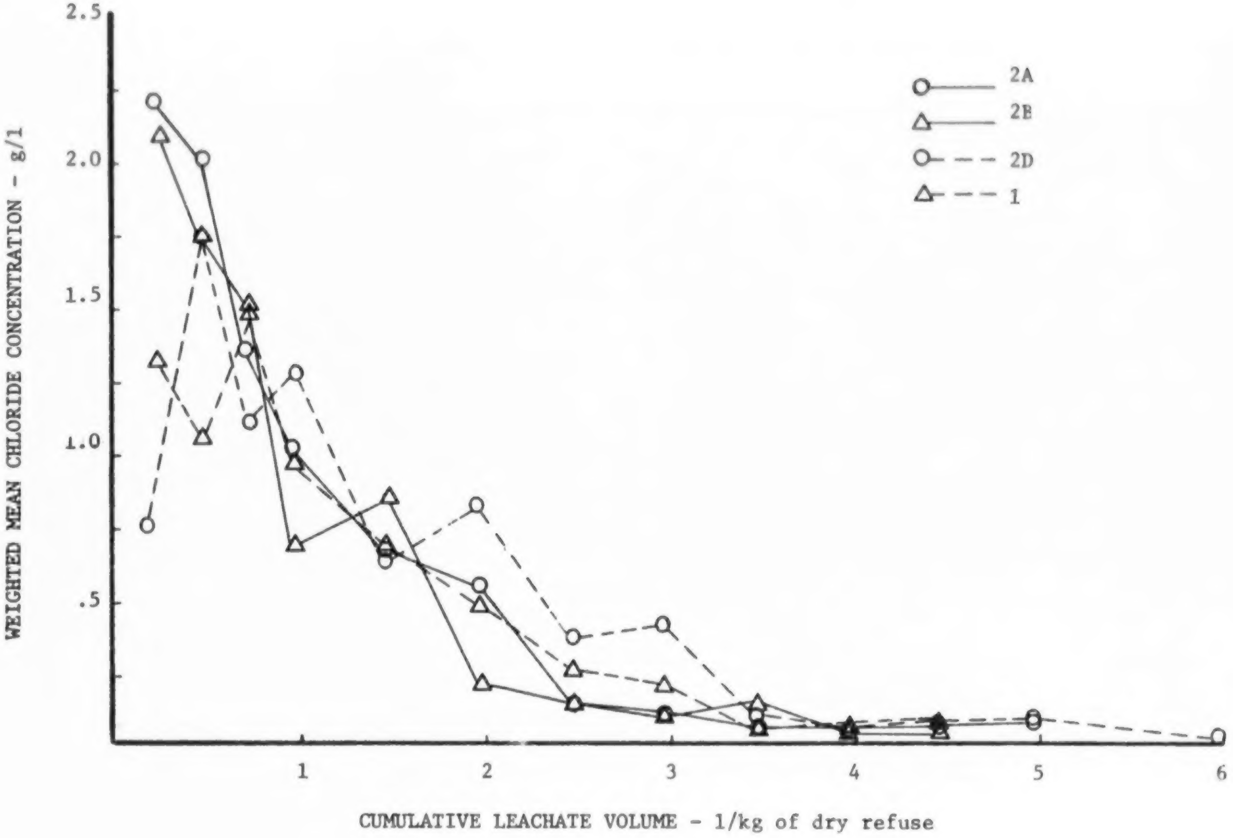


Figure 13. Chloride concentration history

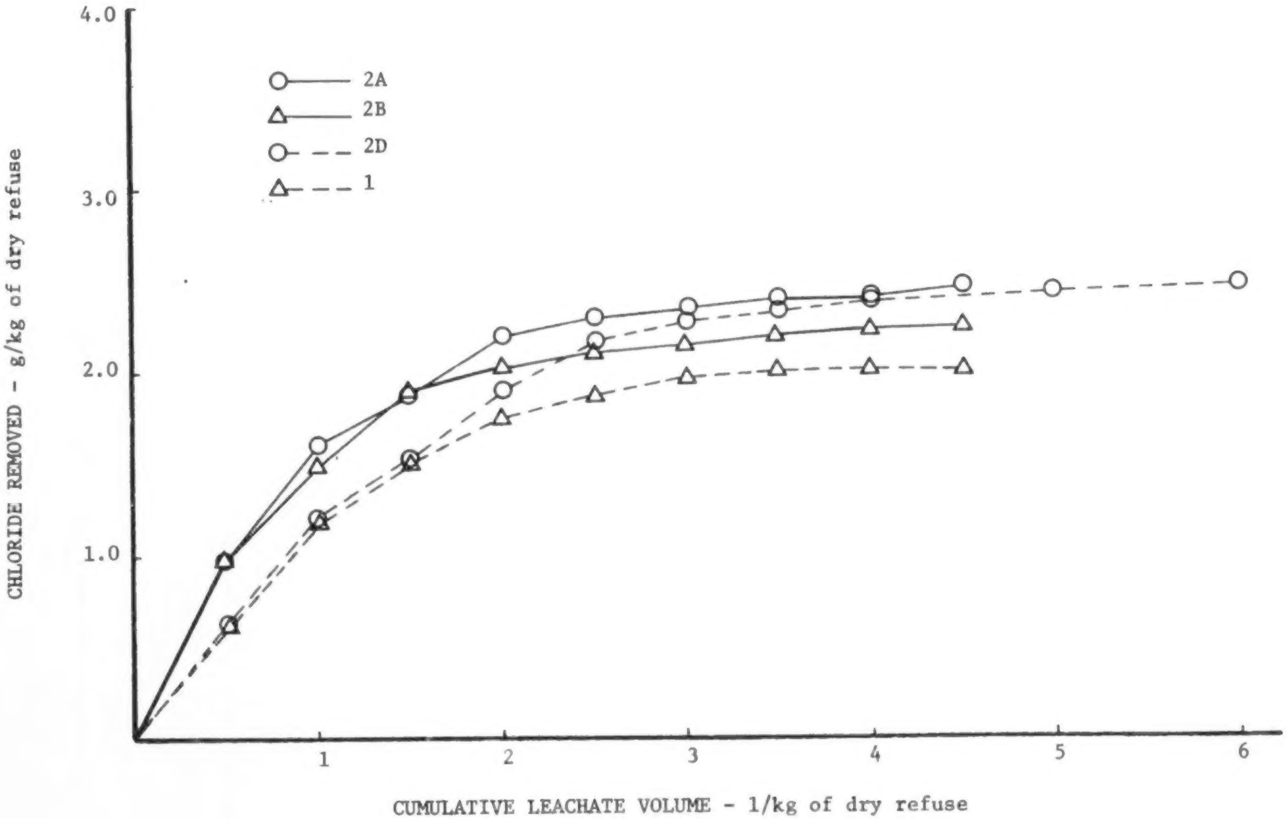


Figure 14. Chloride mass removal

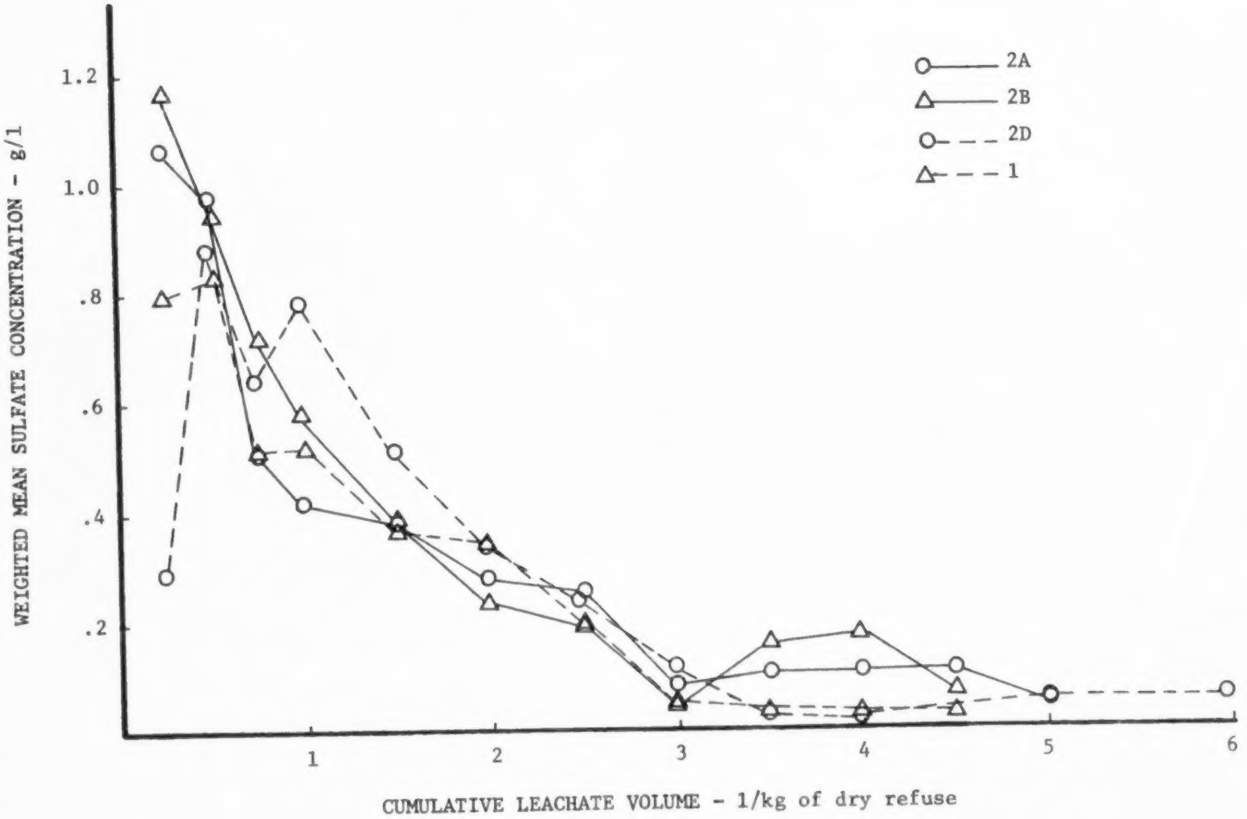


Figure 15. Sulfate concentration history

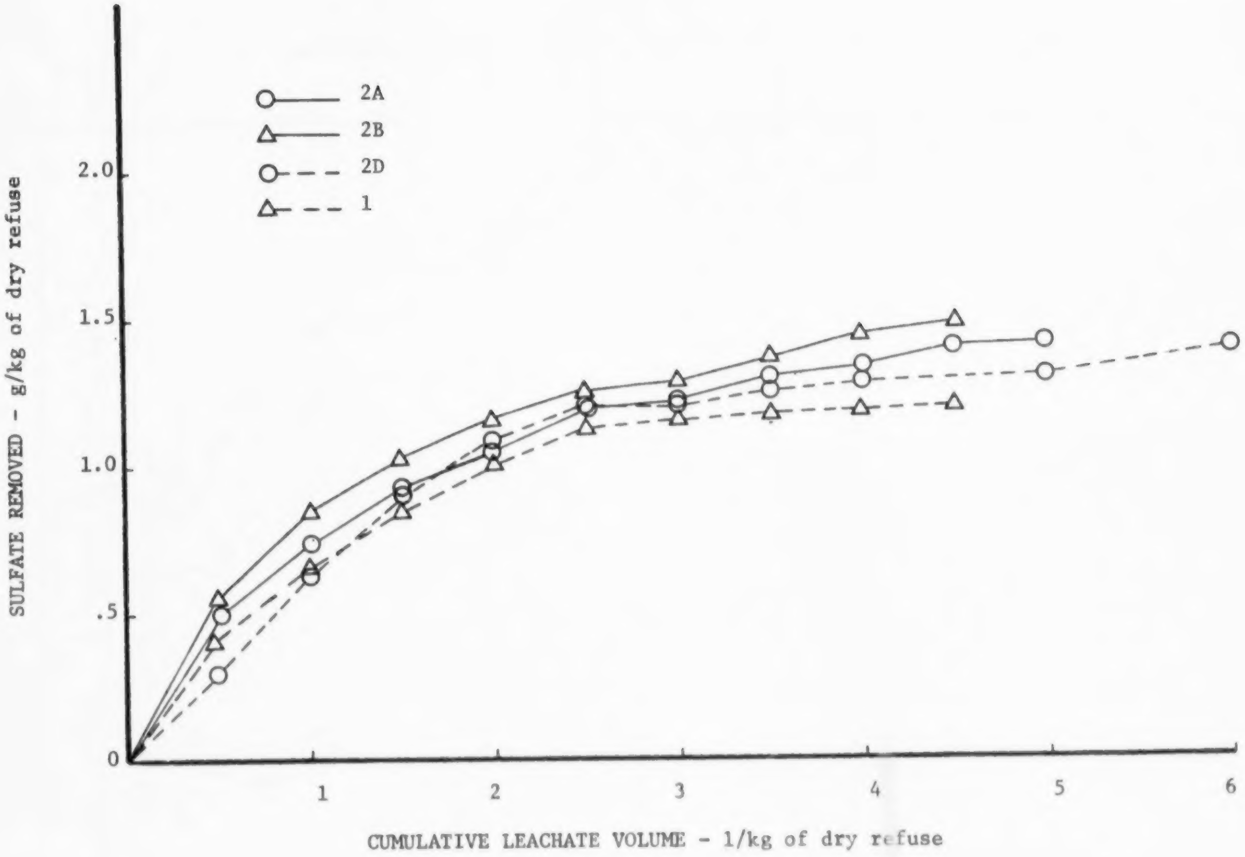


Figure 16. Sulfate mass removal

small-scale cells and the field-scale cell, 2D. It was hoped that the small-scale cells would be adequate models of the large scale cell so that future research efforts might utilize the smaller cells for prediction of field behavior.

It was determined that on the basis of a 10 component analysis of composition that variations in performance between cells 2A, 2B, and 2C could not readily be attributed to variation in refuse composition. Unit densities of 2A, 2B, and 2C were similar, ranging from 392-431 kg/m^3. The refuse depth was the same for all small cells. Water input to each cell was assumed to be the same.

The initial refuse composition for cell 2D was assumed similar to that of the small cells. The density was somewhat greater at 598 kg/m^3 at a slightly higher moisture content. The possible effect on leachate concentration of this greater density is not known. The input of water was controlled the same as for the small-scale cells. Cumulative leachate production was actually greater than the precipitation. Refuse depth was slightly less in 2D at 2.44 m whereas in the small cells the depth was 2.56 m.

For the six parameters presented here, there is a strong similarity in concentration histories for 2A and 2B over the leachate volume range examined. For COD, Cl, sulfate, Mg and hardness, the concentration has dropped in an exponential manner to less than 10% of the peak values recorded. Iron has been reduced to about 50% of peak, indicating a slow release of the mass available for leaching.

Statistical comparison of 2A and 2B concentration histories with an analysis of differences technique (1) was not meaningful. Future efforts for comparison will be attempted with a goodness of fit test, such as the Kolmogorov (3).

Total mass removed plots indicate similar trends for all parameters when comparing 2A and 2B. This would be expected since the leachate volumes and concentration histories were similar. For COD, the field-scale cells cumulative mass removals lag those of 2A and 2B and show very little mass removal after 3 liters/kg of cumulative leachate. Concentrations for these cells have dropped to 1-2% of the highest values. Diluting side wall leakage has probably also contributed to the low mass removals. Other than for Test Cell 1 iron, the cumulative mass removals for the field-scale and the small-scale cells are quite similar, even though the concentrations recorded from the field-scale cells were generally lower in the initial phase of the experiment. The mass removals will also be compared with a goodness of fit test in the future to determine whether the small-scale cells were statistically comparable to the field-scale cells.

MATHEMATICAL DESCRIPTION
OF
LEACHATE CONCENTRATION HISTORY

The repetitive shape of the leachate concentration curves and similar volumes at peak concentrations for many of the parameters indicated that the weighted mean concentration history curves might be mathematically described. The need for such a descriptive model has been documented by Brunner (2). A forecasting technique would provide a significant improvement in the design and regulatory review processes and could be used to determine optimal operational practices. Forecasting would be extremely useful in evaluation and design of leachate treatment needs and the suitability of the hydrogeologic setting.

The concentrations in the leachate are dependent on many factors, not all of which are understood or known, especially the magnitude of influence the factors exert on resultant concentrations. Important variables might be the initial mass of a substance readily available for leaching, decomposition of refuse within the fill and subsequent additional mass availability, the pH, solubility limits, the rate of water throughput, decomposition or reduction of the leachate constituents during travel through the fill, the depth of the fill and cover soil effects within the fill. The number of variables involved and the lack of understanding of their effect on resultant leachate concentrations make a single semi-empirical approach attractive. Previous efforts have been reported by Phelps (6), Straub (7), and Wigh (4).

One option in modeling leachate concentration is to consider the refuse mass as a single well-mixed reactor (7). Such a system is depicted in Figure 17.

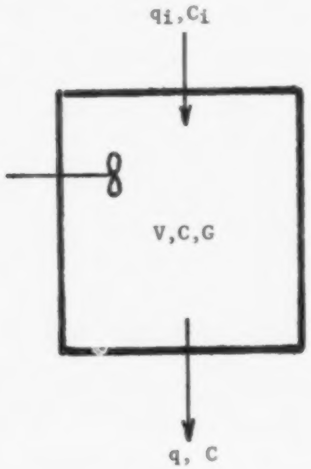


Figure 17. Well-mixed Reactor

C_i and C are the water input and leachate concentrations. q_i and q are water input and leachate flow rates respectively. V is the total water contained in the refuse system and G is the internal mass generation rate.

Writing a mass balance on the reactor yields:

$$\frac{dVC}{dt} = q_iC_i - qC + G \quad (1)$$

or the time rate of change of mass within the system equals mass inflow less outflow plus internal generation. This can be further simplified by assuming that the incoming concentrations are negligible, and that the volume of water within the system is constant or $q_i = q$ and V is a constant. This yields:

$$\frac{VdC}{dt} = qC + G \quad (2)$$

To solve the differential equation, an expression for G is necessary. One option is to consider the mass generation as a first order reaction as is commonly done for biochemical decomposition (8). This expression is simply G = kVC, or the rate of mass generation is proportional to the amount of mass present, with k being the rate coefficient. The differential equation becomes:

$$\frac{VdC}{dt} = qC + kVC \quad (3)$$

which when solved for $C=C_o$ at t=o yields:

$$C = C_oe^{-(q/V - k)t} \quad (4)$$

Thus, a general expression for the concentration change with time with t=o at field capacity is a simple exponential dependent on the initial concentration (C_o), the leachate flow rate q, total volume of water in the refuse at field capacity V, and an empirical rate constant k.

For most of the parameters studied, peak concentrations were recorded when field capacity was being achieved. C_o was therefore selected as that peak concentration for each parameter. V can be calculated for each test cell from initial moisture contents and field capacity determinations. The leachate flow rate (q) must be constant with time to be valid in the equation. This was determined by plotting leachate collected with time as shown in Figure 18. The resultant constants for each test cell are listed in Table 5.

TABLE 5. TEST CELL CONSTANTS

Test Cell	q*	V#
1	.58	.75
2A	.58	1.15
2B	.51	1.04
2D	.67	.92

* Units are liters/kg dry refuse/year
# Units are liters/kg dry refuse

The concentration history curves for 2A and 2B as compared to the general expression with k selected for best visual fit are shown in Figures 19-24. Reasonably good fit was obtained for all the parameters except iron. It is interesting to note that the mass transfer rate coefficient (k) for chloride was zero, indicating that all mass present in the refuse was available for leaching initially or so readily removeable that it was independent of biochemical reactions within the test cells. Magnesium and sulfate had low constants (.15) and iron the highest (.4). Hardness had a rate coefficient of .25, perhaps influenced by the slow iron removal rate.

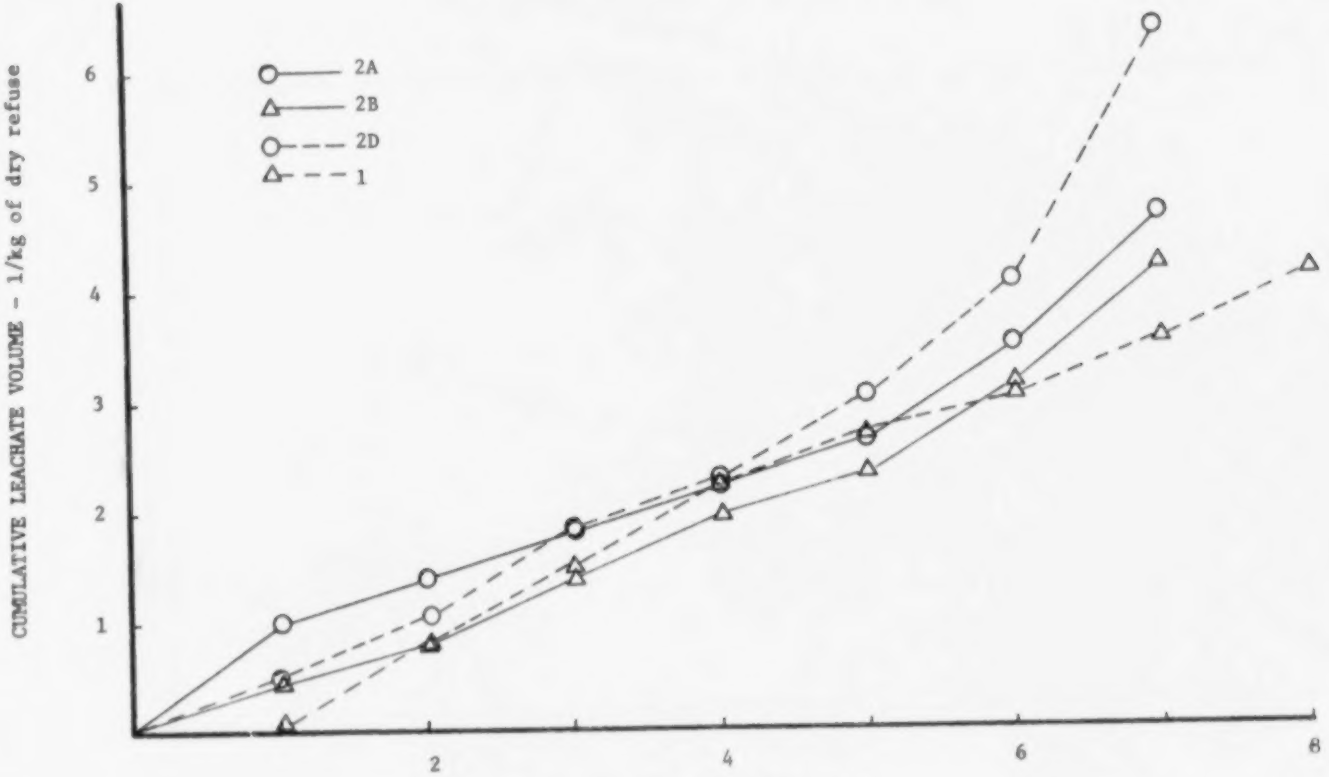


Figure 18. Leachate flow rates

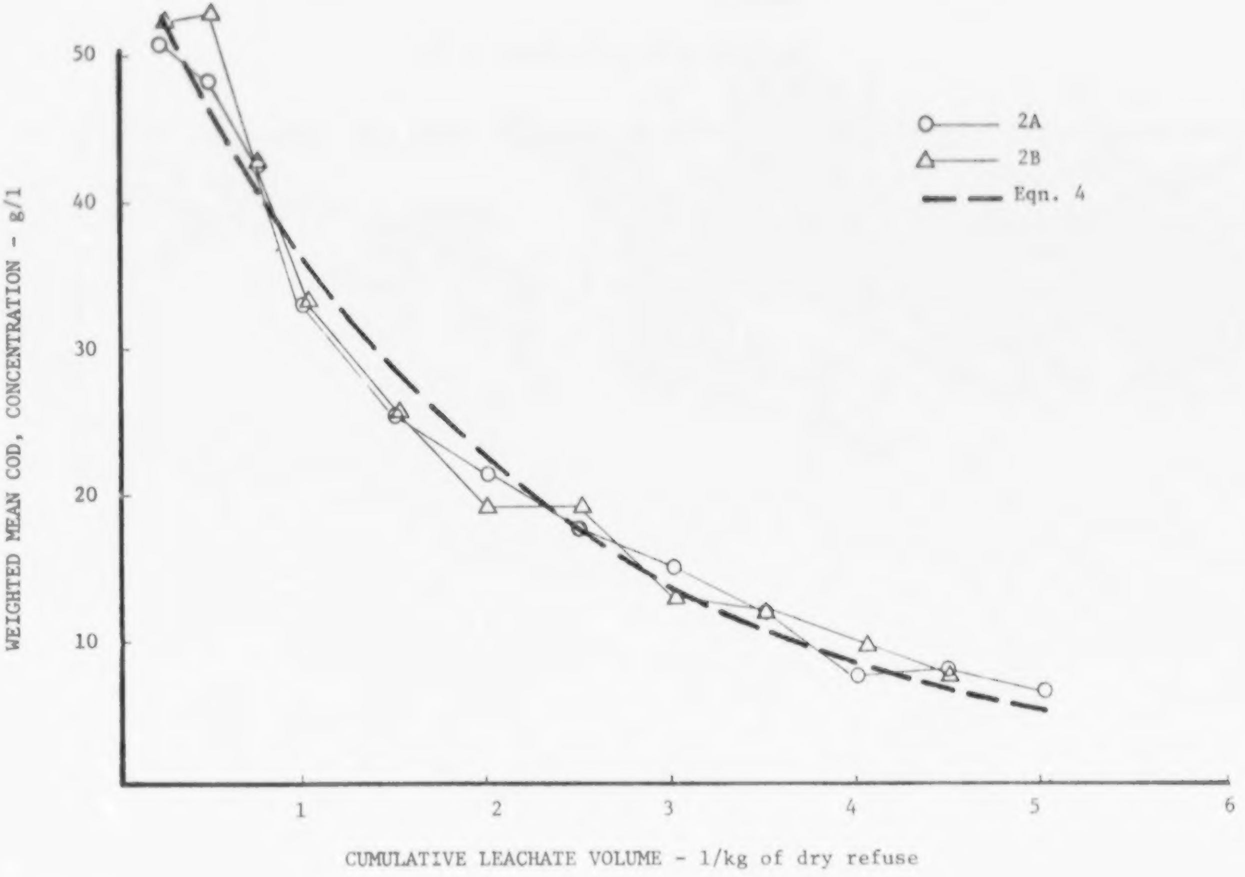


Figure 19. COD Concentration comparison.

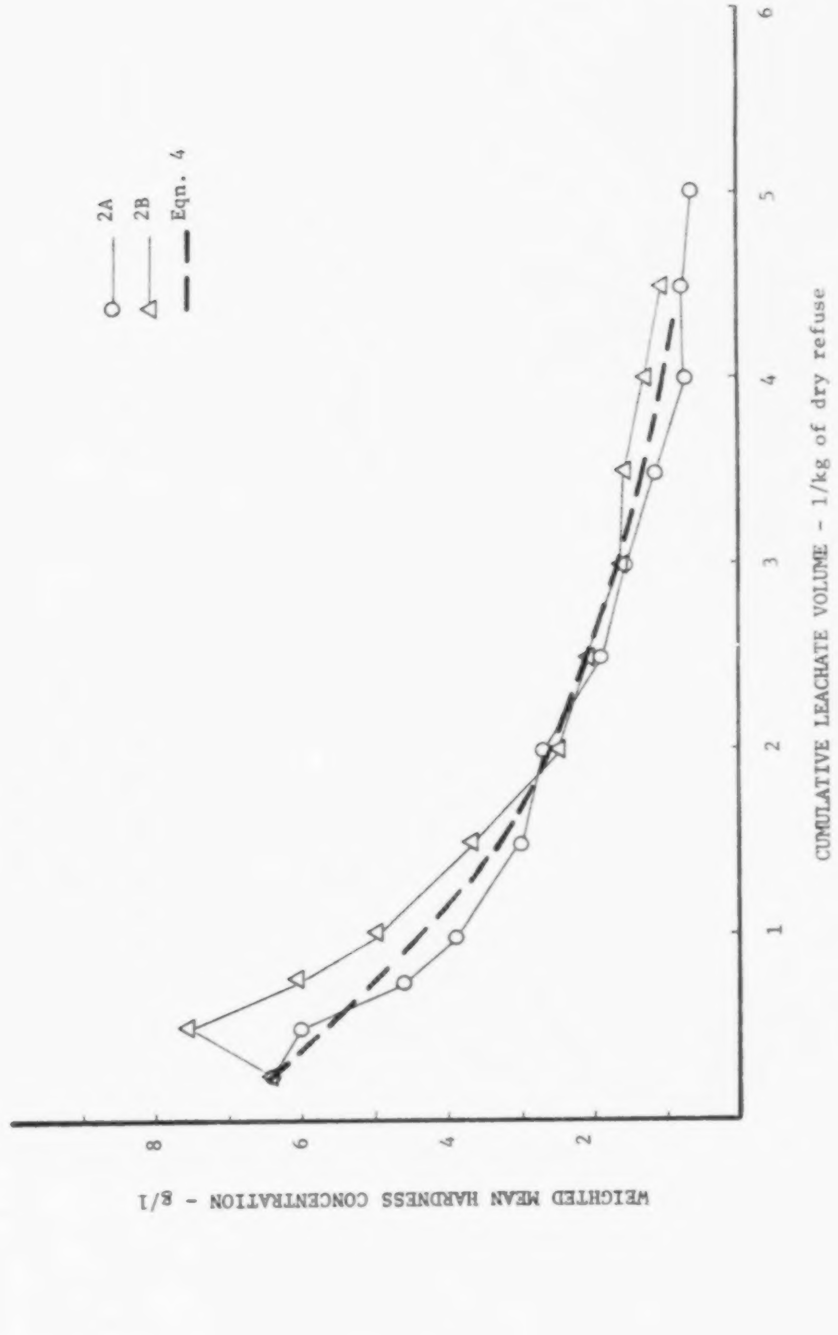


Figure 20. Hardness concentration comparison.

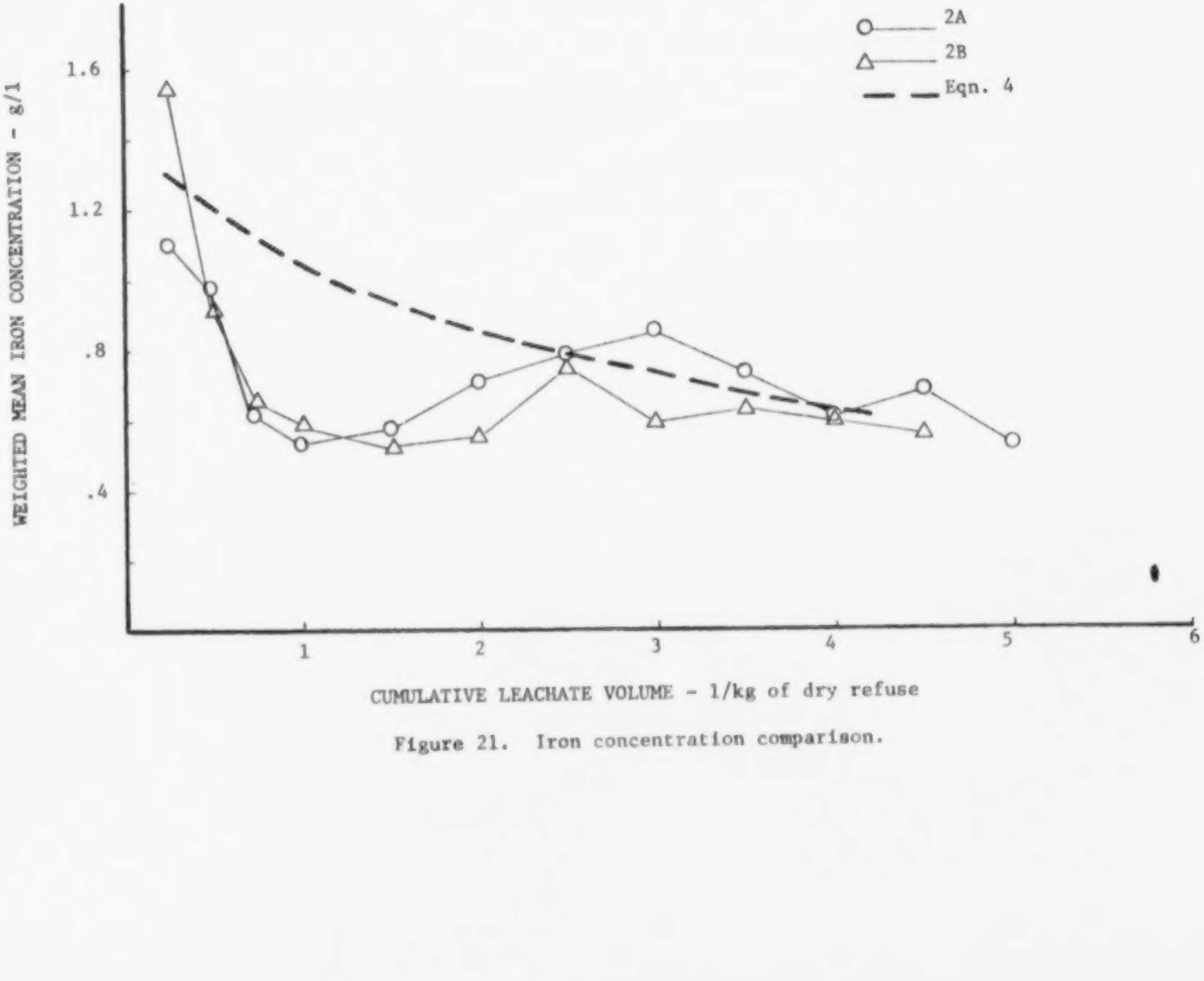


Figure 21. Iron concentration comparison.

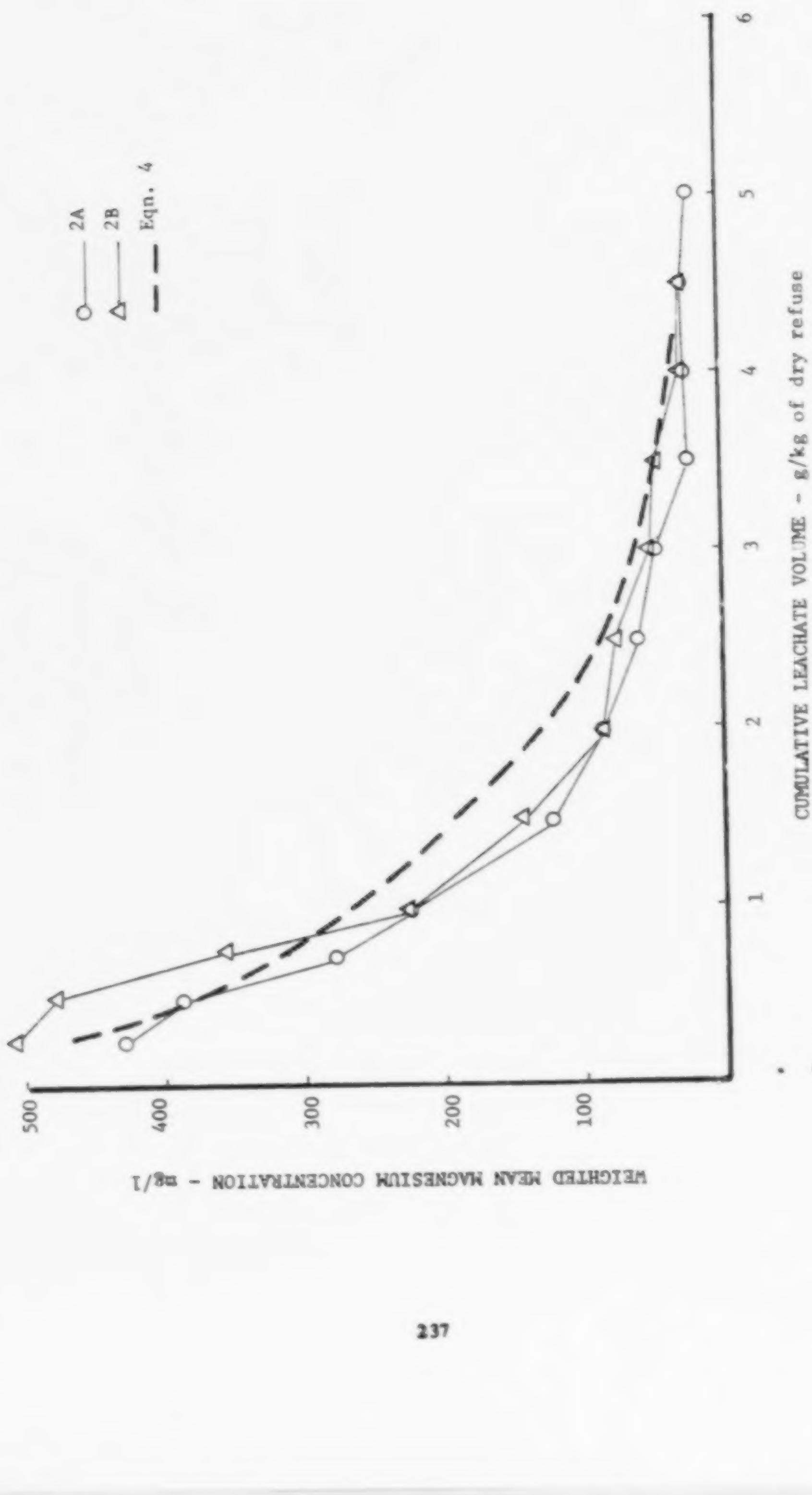


Figure 22. Magnesium concentration comparison.

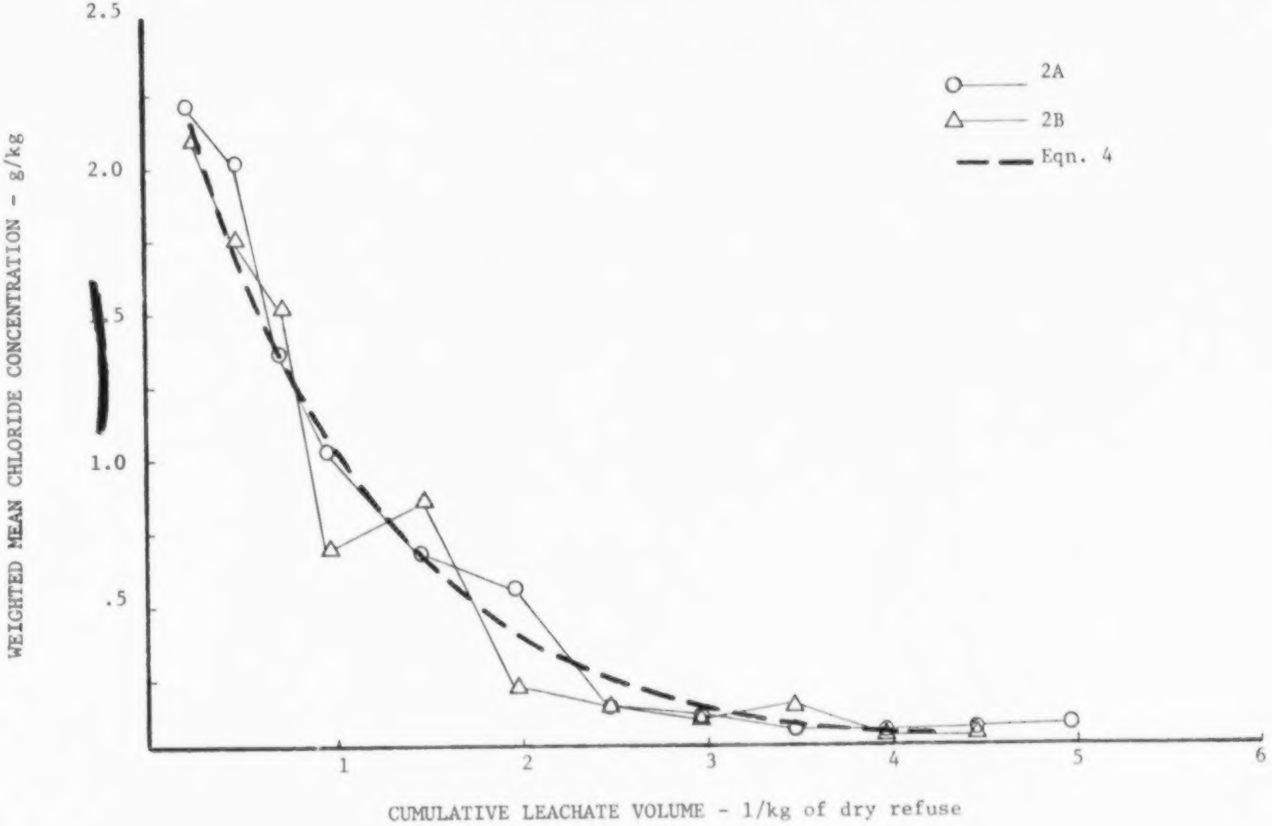


Figure 23. Chloride concentration comparison.

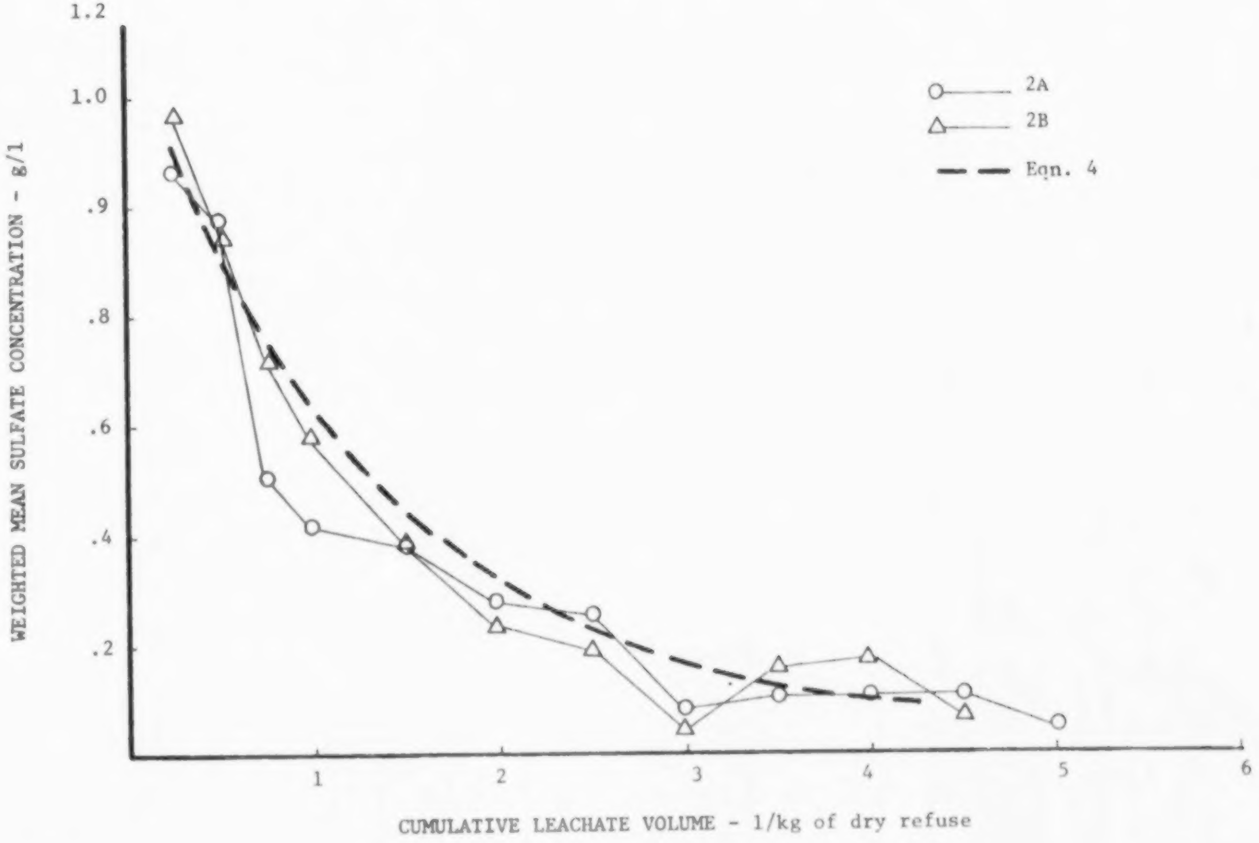


Figure 24. Sulfate concentration comparison.

Mass Removal

Integrating the concentration times the flow rate over infinite time (Cqdt) yields a general expression for total leachable mass.

$$M = \frac{qC_o}{q/V - k} \qquad (5)$$

The total mass available estimated from this expression for 2A and 2B and the amount removed at 4.5 l/kg of dry refuse are shown in Table 6. There is reasonably good agreement between these values and the shape of the mass removal curves.

TABLE 6. 2A-2B MASS REMOVALS AT 4.5 l/kg OF LEACHATE

Parameter	k	Total * Est. Mass	Mass Removed	
			2A	2B
COD	.25	104	96.6	97.8
Hardness	.25	12.8	11.2	13.4
Iron	.40	6.6	3.3	3.0
Magnesium	.15	.7	.52	.62
Chloride	0	2.1	2.5	2.2
Sulfate	.15	1.6	1.4	1.5

* q/kg of dry refuse

As was noted earlier, the peak concentrations of 2D at field capacity were lower than those of 2A-2B. Likely causes were thought to be diluting sidewall leakage and short-circuiting within the cell. Another explanation would be the higher flow rate, less contact time, resulting in reduced concentrations. If the total available masses/unit weight for each cell are considered equal and the mass transfer constants do not change for the higher flow rates then the ratio of the peak concentrations for 2A-2B and 2D can be obtained using Equation 5. The results are shown in Table 7.

TABLE 7. 2D CONCENTRATIONS (mg/l) AT FIELD CAPACITY

Parameter	Est. Conc.	Actual Conc.
COD	33,000	34,500
Hardness	4,030	5,000
Iron	490	880
Magnesium	340	330
Chloride	1,760	1,730
Sulfate	810	870

Application to Field Situation

It must be emphasized that Equation 4 is only applicable to batch-type systems. If, after further comparison to test cells of more variable conditions there is still reasonable agreement, the general expression could possibly be applicable to field conditions. Shown in Figure 25. are two concentration curves, one for the single cell or batch system, and the other the estimated COD concentration for a single lift landfill operated for 10 years. If proven valid by field studies, an expression of this type would provide a valuable tool for siting, design and operation of landfills.

SUMMARY AND CONCLUSIONS

Five sanitary landfill test cells were constructed at the Boone County Field Site during 1971-72. Three of these were small-scale cells containing approximately 2100 kg of dry refuse. The two field-scale cells, 1 and 2D, contained 286,000 and 42,450 kg of dry refuse. The test cells were closed in August, 1980.

Preliminary findings from analysis of part of the data include:

1. Leachate collected from Test Cell 1 was 27% of precipitation after 6.5 years. Water balance calculations were reasonably accurate in predicting the quantity of leachate, provided actual climatic data rather than average was used. This demonstrates the need for analyzing a variety of

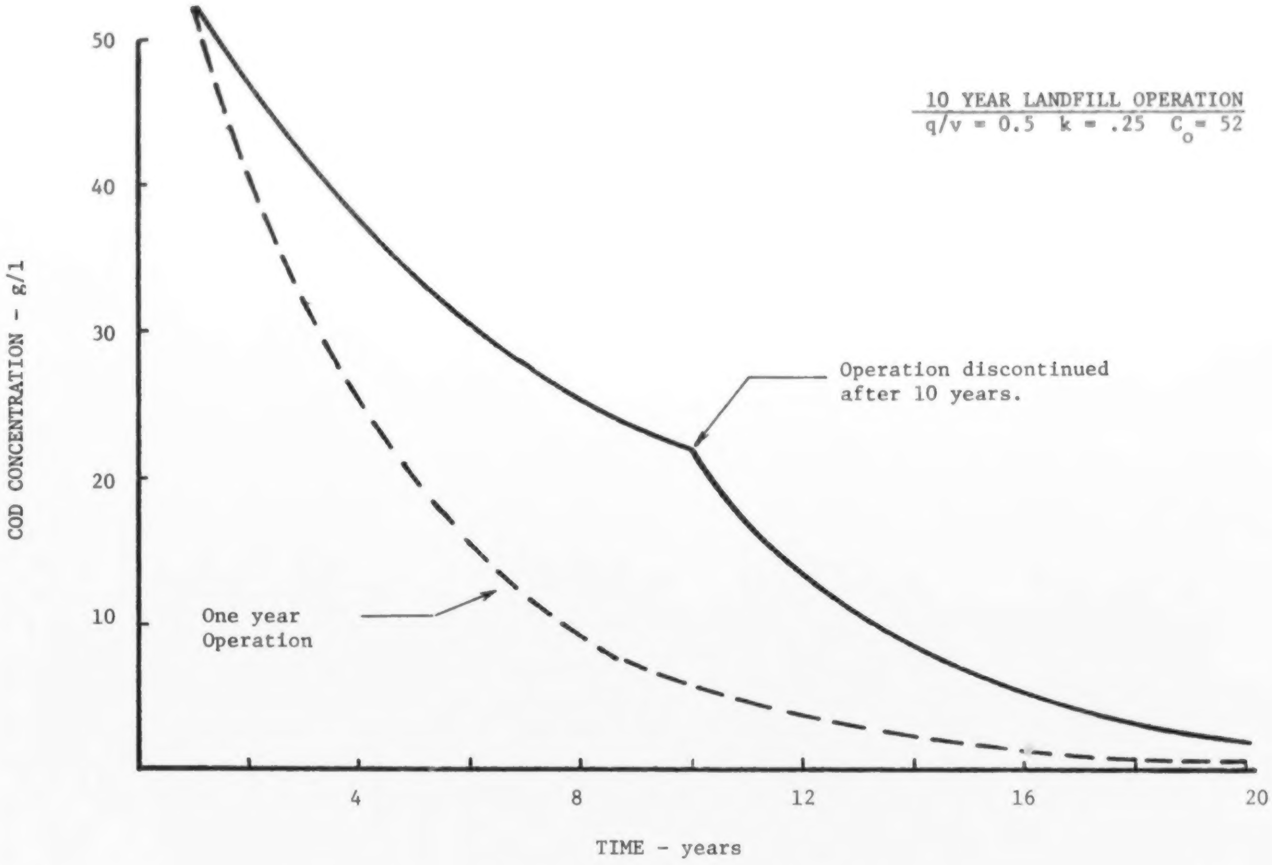


Figure 25. Projected landfill leachate COD concentrations.

weather conditions when using the water balance method as a design tool.

2. The efficiency of the soil liner in Test Cell #1 could not be evaluated because of interference from the underlying synthetic liner.

3. Leachate concentration histories showed similar trends for all cells, with most parameter peaks occuring at field capacity. After 4-5 l/kg of refuse of leachate had been collected, contaminant concentrations were generally less than 10% of peak values.

4. From 56-104 grams of COD, 11-13 grams of hardness, 1.2-1.5 grams of sulfate, .52-.62 grams of magnesium, 2.0-2.4 grams of chloride and 1.9-3.6 grams of iron per kg of dry refuse had been leached from the cells at the close of the study. Mass removal rate trends for all but iron were tending towards no further removal.

5. A simple exponential equation provided an adequate description of the concentration histories of 5 of the 6 parameters examined. Comparison of predicted total mass removals with actual mass removals showed reasonable agreement. The theory should be applied to other test cells and field conditions to verify applicability.

ACKNOWLEDGMENT

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